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5 **Protecting an ecosystem service: approaches to understanding and mitigating**
6 **threats to wild insect pollinators**

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Contents

1. Importance of Insect Pollination	3
1.1 Providing an Ecosystem Service	3
1.2 Brief Introduction to Pollination Ecology and the Importance of Wild Pollinators	8
2. Major Threats to the Pollination Service Provided by Insects	13
3. Steps in the Right Direction to Protect Insect Pollinator Services: Policy Actions	19
4. Understanding and Mitigating Specific Threats to Wild Insect Pollinators to Protect Pollinator Services	21
4.1 Understanding the Stability of Insect Pollinator Communities	22
4.2 Using Molecular Approaches to Monitor Insect Pollinators	24
4.3 How Do Parasites Shape Wild Insect Pollinator Populations?	29
4.4 Understanding Insect Pollinator Population Responses to Resource Availability	31
4.5 Engineering Flowering Field Margins as Habitats to Attract Insect Pollinators	35
4.6 How Might We Improve the Wider Countryside to Support Insect Pollinators	39
4.7 Insect Pollinators in Urban Areas	41
5. Considerations When Developing Future Research and Mitigation Strategies	45
Acknowledgements	47
Appendix	48
References	52

72 **Abstract**

73

74 Insect pollination constitutes an ecosystem service of global importance, providing
75 significant economic and aesthetic benefits as well as cultural value to human society,
76 alongside vital ecological processes in terrestrial ecosystems. It is therefore important to
77 understand how insect pollinator populations and communities respond to rapidly changing
78 environments if we are to maintain healthy and effective pollinator services. This paper
79 considers the importance of conserving pollinator diversity to maintain a suite of functional
80 traits to provide a diverse set of pollinator services. We explore how we can better understand
81 and mitigate the factors that threaten insect pollinator richness, placing our discussion within
82 the context of populations in predominantly agricultural landscapes in addition to urban
83 environments. We highlight a selection of important evidence gaps, with a number of
84 complementary research steps that can be taken to better understand: i) the stability of
85 pollinator communities in different landscapes in order to provide diverse pollinator services;
86 ii) how we can study the drivers of population change to mitigate the effects and support
87 stable sources of pollinator services; and, iii) how we can manage habitats in complex
88 landscapes to support insect pollinators and provide sustainable pollinator services for the
89 future. We advocate a collaborative effort to gain higher quality abundance data to
90 understand the stability of pollinator populations and predict future trends. In addition, for
91 effective mitigation strategies to be adopted, researchers need to conduct rigorous field-
92 testing of outcomes under different landscape settings, acknowledge the needs of end-users
93 when developing research proposals and consider effective methods of knowledge transfer to
94 ensure effective uptake of actions.

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96

97 **1. Importance of insect pollination**

98

99 **1.1 Providing an ecosystem service**

100

101 Insects show an astonishing taxonomic diversity and are abundant in almost all
102 environments across the globe (ca. 1 million described species, with an estimated >4 million
103 undescribed; Stork et al., 2015). Their importance for the functioning of ecosystems cannot
104 be overstated, contributing to fundamental ecosystem processes including soil turnover,
105 decomposition and nutrient cycling, and play key roles in local food webs (Schwartz et al.,
106 2000). These processes have direct implications for human welfare by providing critical
107 ‘ecosystem services’, which were defined by Fisher and colleagues as “*the aspects of*
108 *ecosystems utilized (actively or passively) to produce human well-being*” (Fisher et al., 2009:
109 modified from Boyd and Banzhaf 2007). For example, predatory and parasitoid insects play
110 the role of natural enemies of pest herbivores and so help to control the pests of agricultural
111 crops grown to feed the human population.

112

113 Of the multiple roles that insects play, pollinating flowering plants is a process that is
114 of the utmost importance in terrestrial environments and one which provides vital ecosystem
115 services for human wellbeing (Carpenter et al., 2006; Garibaldi et al., 2011a, 2014). Over
116 85% of described flowering plant species are dependent, to some degree, on animal
117 pollination (Ollerton et al., 2011), with mobile foraging insects accounting for the vast
118 majority of this activity (Kremen et al., 2007). Insect pollination maintains genetic diversity
119 in plant populations (Kearns et al., 1998), and provides advantages such as increased fruit
120 quality and quantity, and seed production and fertility, leading to greater vigour of the next
121 generation (Albrecht et al., 2012; Barrett, 2003). Thus, ecologically, insect pollination is

122 crucial in sustaining diverse and healthy populations of wild plants and this in turn underpins
123 trophic networks (Memmott, 1999). Insect pollination also has high economic value to
124 humans (Allsopp et al., 2008, Hein, 2009; Palomo et al., 2015; Winfree et al., 2011b),
125 significantly increasing the yield and health of cultivated crops (see Table 1), placing the
126 value of insect pollination for world agriculture > €150bn (Gallai et al., 2009; Lautenbach et
127 al., 2012), as well as being of high aesthetic and cultural value through supporting florally
128 diverse landscapes (Lovell and Sullivan, 2006; Wratten et al., 2012).

129

130 A growing global human population (9bn by 2050) and increases in the average
131 calorific intake per person, places rising demands for food security under rapidly changing
132 environments (Godfray et al., 2010). As much as 75% of agricultural crop species are, to
133 some degree, reliant on animal pollination (Table 1), with the suggestion that a complete loss
134 of this service could reduce yields by ca. 40% and 16% for fruit and vegetables, respectively
135 (Klein et al., 2007). Furthermore, most lipids and micronutrients important for the human diet
136 and public health are obtained from plants requiring animal pollination (Chaplin-Kramer et
137 al., 2014; Eilers et al., 2011). With insects constituting the vast majority of animal
138 pollination, we should recognise our duty to help maintain sustainable and healthy insect
139 pollinator populations and take full advantage of this ‘free’ ecosystem service effectively.
140 Reports of insect pollinator declines in many parts of the world suggest, however, that we
141 may not have lived-up to this responsibility (see section 2) increasing the risk of future
142 pollination deficits in areas requiring high, and increasing, pollination demands (Aizen et al.,
143 2008a; Aizen and Harder, 2009; Lautenbach et al., 2012; Polce et al., 2014).

144

145 Areas experiencing drastic transitions in land-use from natural habitats to large scale
146 intensive agriculture, may find that native resident pollinators are unable to provide the level

147 **Table 1.** Selected recent studies showing the effects of wild or managed pollinators on commercially important crops. Studies consist of reviews
 148 of primary literature that present an overview of the global picture (1-3) and recent studies showing experimental manipulations that did not
 149 focus exclusively on honeybees (4-9). Pollinators were either allowed open access to inflorescence (OP) or excluded while still allowing wind
 150 and self-pollination (WS) to occur. The studies were chosen as examples and the table is not an exhaustive coverage of the literature.

	Major insect pollinator(s)	Plant host(s)	Quality measure(s)	Description	Authors
1	Various	Various	Various	Review of 252 crop species showing that majority of important crops benefit from animal (overwhelmingly insect) pollination by increasing either seed production, fruit set or fruit number	(Klein et al., 2007)
2	<i>Apis mellifera</i> , Various	Various	Fruit set	Study of 41 global crop systems, showing general trend for i) fruit set to increase with visitations from wild pollinators, ii) magnitude of increase in fruit set to be significantly greater for plants visited by wild pollinators relative to honeybees.	(Garibaldi et al., 2013)
3	Various	Various	Economic impact	Review of 90 studies of crop visiting pollinators using data from 1394 global field sites, concluding that crop visits from bees provided pollination service that contributed ca. \$3,251ha ⁻¹	(Kleijn et al., 2015)
4	<i>Apis mellifera</i> , <i>Bombus spp.</i> Hoverflies	Oilseed rape (<i>Brassica napus</i>)	Seed set, seed quality, market value	Experimental manipulation using ten spring oilseed rape (canola) fields that were either OP or WS, showing OP comparatively increased seed weight and quality but had no effect on seed set (estimated increase in market value of crop by ca. 20%).	(Bommarco et al., 2012b)
5	Various	Gala & Cox Apple (<i>Malus domestica</i>)	Fruit set, seed set, fruit quality, economic impact	Experimental manipulation of pollination across six orchards in two commercial apple varieties. OP treatment produced significant increase in fruit set and seed number compared to WS (economic benefits ca. £11,900 and £14,800ha ⁻¹ for Cox and Gala varieties respectively).	(Garratt et al., 2014a)
6	<i>Apis mellifera</i> , Wild bees Hoverflies	Blueberry (<i>V. corymbosum</i>)	Fruit set, seed set, fruit weight	Experimental manipulation using ten paired sites with either wildflower planted (WF) or mowed field (MF) margins across four years. WF increased fruit set, average berry weight and seed set but for only 3 of 4 years after planting; increase corresponded to increase in abundance of wild bees and hoverflies in same years.	(Blaauw and Isaacs, 2014a)
7	<i>Apis mellifera</i> , <i>Osmia bicornis</i>	Strawberry (<i>Fragaria x ananass</i>)	Fruit set, fruit weight, Economic value	Experimental manipulation of ten plots in areas with established communities of commercially available pollinators. Inflorescences receiving OP produced higher prop. of marketable fruit, higher fruit weight and longer shelf life compared with WS (increased commercial value per fruit by 38.6%).	(Klatt et al., 2014)
8	<i>Apis mellifera</i> , <i>Bombus spp.</i>	Field bean (<i>Vicia faba</i>)	Seed set, Seed weight	Experimental manipulation at 10 sites, with OP produced twice the seed and contained beans that were approximately 9% heavier than the WE treatment	(Nayak et al., 2015)
9	<i>Apis mellifera</i> Various	Apple (<i>M. domestica</i>)	Fruit set	Study of 47 orchards with managed honeybee colonies either present or absent, reporting proportion of fruit set was higher in OP compared to WS. Fruit set was not significantly affected by honeybee presence but did increase with wild bee species richness.	Mallinger and Gratton (2015)

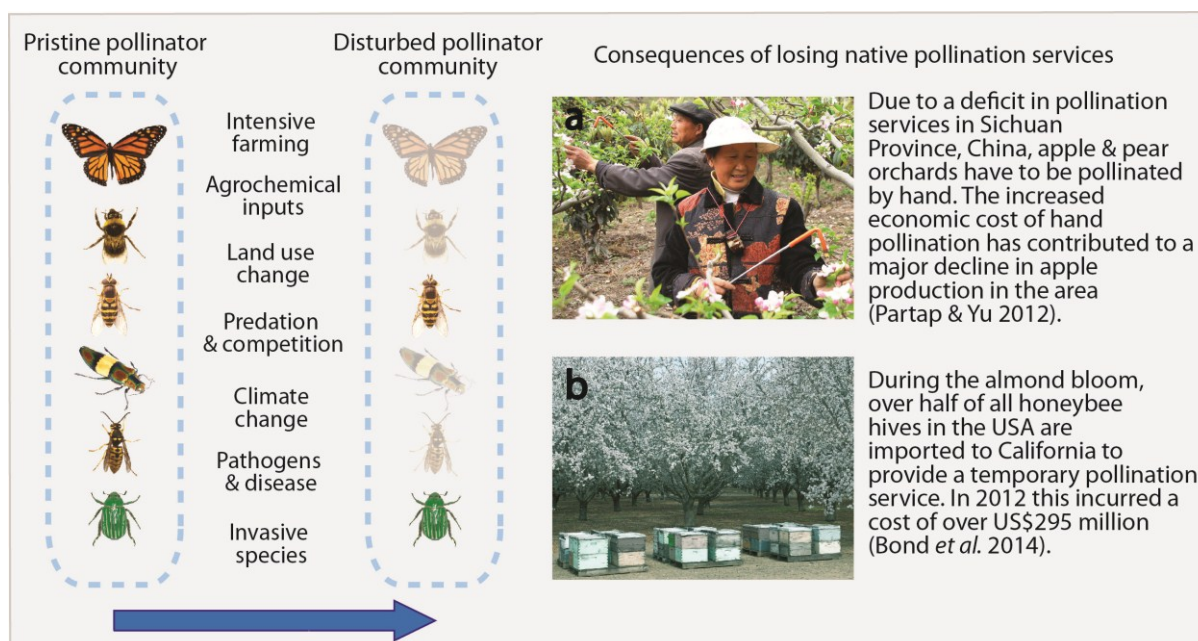
151 of pollination service required (Astegiano et al., 2015; Klein et al., 2012; Kremen et al., 2002,
152 2004). For example, in the state of California, USA, half of the country's domesticated
153 honeybee hives are transported to the region to pollinate large scale cultivations (such as
154 Almonds: Aizen and Harder, 2009), a tactic similarly adopted by other countries across the
155 globe (Crane, 1990; Klein et al., 2007; Palomo et al., 2015). In the apple and pear orchards of
156 the Sichuan province of China, apparent reductions in wild bee populations, have led to a
157 dependency on flowers having to be pollinated by hand (Figure 1; Partap & Partap, 2002;
158 Partap and Ya, 2012). Instead of receiving a free and rapid pollination service from a healthy
159 insect community (Morandin and Winston, 2006), it has been replaced by comparatively
160 expensive human labour. Apple and pear cultivation in the area has now declined
161 significantly (Partap and Ya, 2012), and it thus seems that in the long-term, crops which are
162 less dependent on insect pollinators (such as plum, loquat, walnut, and other cereal crops),
163 will be grown instead. Such action may only exacerbate the problem to local insect pollinator
164 populations as this could further reduce nutritional resource availability in those landscapes,
165 especially as many self-compatible plants produce low (or zero) nectar content and can
166 produce less nutritious pollen that may also be harder for insects to collect (Zimmerman,
167 1988; Johnson & Bond, 1997).

168

169 **1.2 Brief introduction to pollination ecology and the importance of wild pollinators**

170

171 Many angiosperm species possess relatively open flower morphologies, allowing a
172 range of insect pollinator species access to the nectar and pollen rewards within. Such plant-
173 pollinator mutualisms are often generalised involving many species interactions (Willmer,
174 2011). Whilst the dynamics of mutualistic networks can be relatively complex, we can make
175 the broad theoretical prediction that generalist species may be more resistant to



176

177 **Figure 1|** Combined and interactive set of environmental stressors can threaten pollinator
 178 communities. Persistently induced stress may lead to pollinator population declines resulting
 179 in the composition and stability of pollinator communities to be affected with potential loss of
 180 species richness and reduced pollination insurance. Loss of pollinator services can impact on
 181 human welfare in many ways, but a major concern is that native resident pollinator
 182 communities are no longer able to keep-up and meet demands leading to ineffective
 183 pollination of agricultural crops and wildflowers. For example, inadequate local pollination
 184 services have led to the implementation of extreme measures, including costly replacement
 185 by: a) employing human labour to hand pollinate flowers (Partap and Yu, 2012; photo taken
 186 and provided with permission by Uma Partap), or b) manage domesticated pollinators such as
 187 honeybees (Bond et al., 2014). Many insect taxa constitute a pollinator community all of
 188 which visit flowers for nectar and pollen rewards, but certain Orders constitute the vast
 189 majority of insects providing a pollination service: the Coleoptera (beetles), Diptera (flies),
 190 some Hemiptera (true bugs), Hymenoptera (bees, many wasps & some ants), Lepidoptera
 191 (butterflies & moths) and Thysanoptera (thrips).

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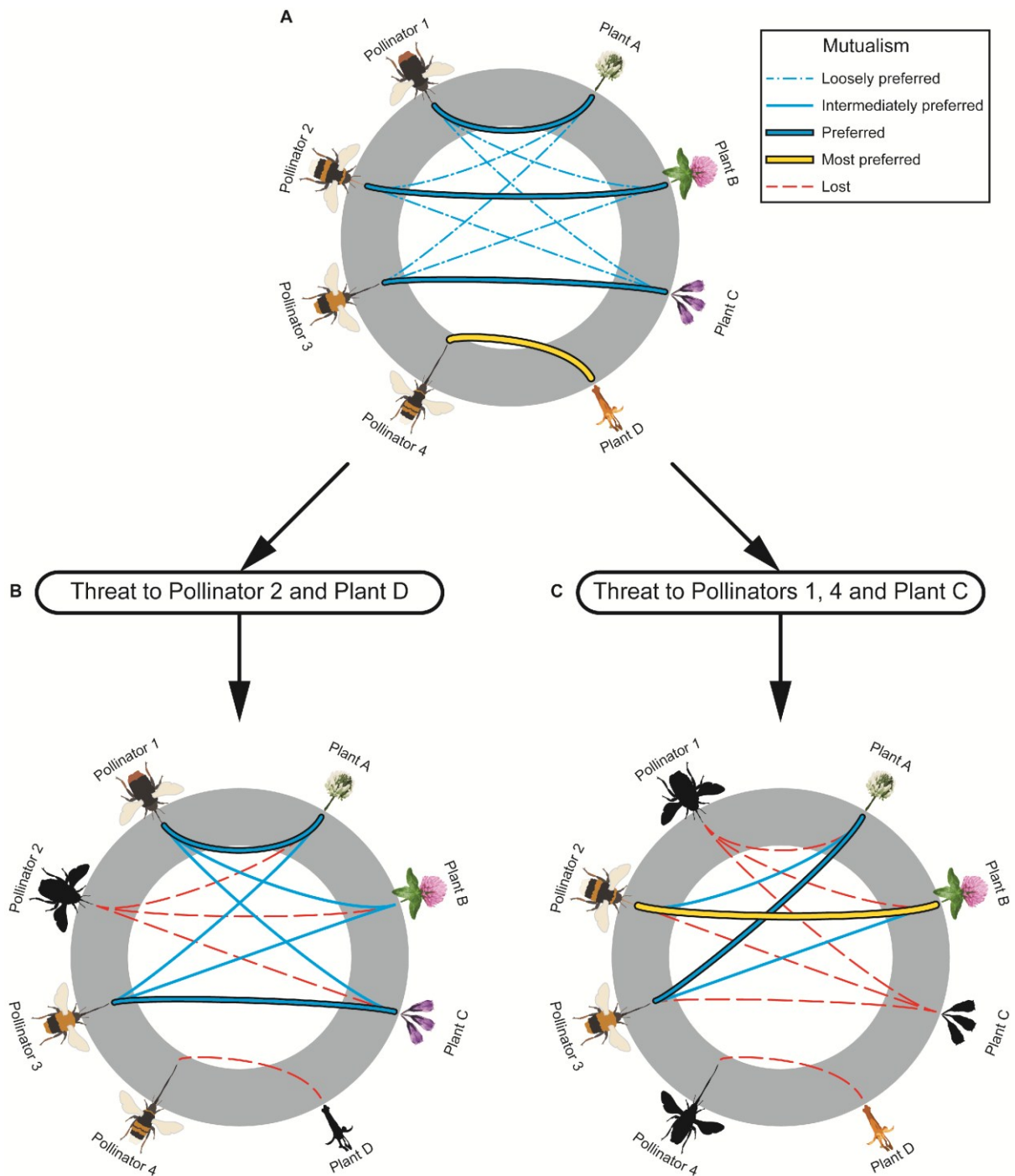
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196 environmental perturbations, depending on its extent, and their presence may contribute to
197 greater redundancy in pollination function and community resilience to global change
198 (Astegiano et al., 2015; Lever et al., 2014). For example, if environmental changes extirpate
199 or reduce the abundance of the primary pollinator of a plant, or vice versa, then other
200 generalist species depending on their functional traits may be able to fulfil the role of
201 mutualist by engaging in flower visits (Figure 2; Aizen et al., 2012; Ashworth et al., 2015;
202 Burkle et al., 2013; Memmott et al., 2004; Mitchell et al., 2009b; Waser et al., 1996; but see
203 Kaiser-Bunbury et al., 2010; Winfree et al., 2014). Communities of generalist species,
204 sharing similar functional traits, may have high inter-specific competition between pollinators
205 for shared floral resources or between plants for pollinator functional groups (Johnson and
206 Steiner, 2000; Mitchell et al., 2009a; Rosas-Guerrero et al., 2014; Figure 2), especially under
207 scenarios of declining wildflower or pollinator populations (see sections 2 & 4). There is,
208 however, some evidence from empirical data coupled with simulation modelling that
209 increased network nestedness (high fraction of shared interactions) may facilitate an increase
210 in the number of species constituting the community (Bastolla et al., 2009; Lever et al.,
211 2014). In contrast, some plant species have floral traits that have co-evolved to be
212 specifically adapted to the feeding apparatus of one or a minority of insect pollinator species,
213 thereby representing a relatively exclusive mutualism. For example, the secretion of nectar at
214 the base of a deep corolla or spur is associated with visitation by long-tongued insects (Arditti
215 et al., 2012), or flowers whose anthers require pollen release by vibration (sonication) are
216 pollinated by insects whose thoracic flight muscles can vibrate at an appropriate resonant
217 frequency – process known as buzz pollination (De Luca and Vallejo-Marin, 2013).

218

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221

222 **Figure 2** | Simplified and hypothetical plant-pollinator network, showing a community of four
 223 insect pollinator species and four flowering plant species and the potential consequences of
 224 localised species extinction(s) to network structure. Scenario A) Pollinators 1-3 and plants A-
 225 C show a generalist mutualism, where each pollinator and plant has a preferred relationship
 226 (shown by line thickness and colour), but will also visit / receive other inter-specifics to
 227 subsidise their nectar or pollen intake / pollen transport. The strength of the mutualism may
 228 be dependent on pollinator competition, for instance: pollinator 1 has a preference for plant A

229 and can theoretically competitively exclude (to some degree) pollinators 2 and 3 from visiting
230 plant A (Brosi and Briggs, 2013; Sauve et al., 2014). Pollinator 4 has a longer tongue
231 compared to the other pollinators, and a specialised mutualism with plant D with no
232 competition from the other pollinators as they are unable to reach the nectar reward at the
233 base of plant D's long corolla. Thus, pollinator 4 has a high preference, or dependency, on
234 plant D. Scenario B) Loss of pollinator 2 (i.e. local extinction) changes the preference (or
235 dependency) of the other two pollinators due to competitor release: plant B receives higher
236 visitation rates from pollinator 1 and 3 compared to when pollinator 2 was previously present.
237 This shows that the generalist network has a level of redundancy where other pollinators with
238 similar functional traits (i.e. tongue length) can fill the role of pollinator 2 and maintain a
239 population of plant B in the community an example of network re-wiring. In contrast, loss of
240 plant D is likely to affect pollinator 4 more significantly, and dependent on the ability to visit
241 short corolla flowers, will reduce population size, showing there is lower redundancy in this
242 specialist network. Scenario C). If pollinator 4 were to be lost this would cause localised
243 extinction of plant D. Loss of pollinator 1 will cause competitive release for pollinators 2 and
244 3, but with loss of plant D, it will likely change preferences (or dependency) for plant A and
245 B as well. It will likely strengthen the preference between pollinator 2 and plant B as less
246 floral resources are available. Pollinator 3 will have to change its preference to plant A as it
247 will encounter high competitive exclusion when visiting plant B.

248

249 Having numerous specialist mutualisms within a plant-pollinator community,
250 whereby the plant and pollinator are specialised on each other (and interact with fewer other
251 species) may increase overall functional trait diversity. However, this may result in low
252 functional redundancy at the community network level due to a lack of substitutable species
253 traits or, in other words, there are a small number of pollinators and/or plant species
254 possessing compatible functional traits to fill the functional gap should either mutualist
255 decline (Figure 2; Mouillot et al., 2013; Weiner et al., 2014). The spectrum of interactions
256 spanning generalist to specialist interactions is important in determining how some plant-
257 pollinator relationships remain relatively stable under environmental change, whilst others
258 may not (Burkle and Alarcon, 2011; Burkle et al., 2013; Dalsgaard et al., 2013). Moreover,

259 when floral resources are limited generalist species may gradually replace specialists due to
260 their ability to forage on a range of resources (Carre et al., 2009; Scheper et al., 2014; Waser
261 et al., 1996). At a habitat scale, we would expect maintenance of pollinator species richness
262 to be linked to maintenance of high floral trait diversity, and vice versa. This highlights the
263 importance of conserving bio-diverse habitats to maintain ecosystem functioning and
264 importantly the provision of ecosystem services (Balvanera et al., 2006; Bartomeus et al.,
265 2013; Cardinale et al., 2006; Daily, 1997).

266

267 Bees are frequent and effective pollinators of wildflowers and agricultural crops
268 (Kleijn et al., 2015). Social bees (e.g. honeybees, bumblebees, stingless bees), in particular,
269 have become a focus of insect pollinator studies, in which the numerical abundance of
270 workers per colony make them a dominant ‘pollination service provider’ in many landscapes.
271 The domestication of honeybees has allowed humans to exploit their pollination service by
272 transporting colonies in man-made hives to sites with high pollination demand (Breeze et al.,
273 2011; Potts et al., 2010b; for bumblebee and solitary bee management see: Palomo et al.
274 2015; Pitts-Singer and Cane, 2011; Velthuis and van Doorn, 2006). The significant loss of
275 honeybee colonies each year (Lee et al., 2015; Neumann and Carreck, 2010; Oldroyd, 2007;
276 Potts et al., 2010b), has raised concern, in both the food production industry and the public
277 domain, for the effect on the overall pollination service that honeybees provide (Aizen and
278 Harder, 2009; Polce et al., 2014).

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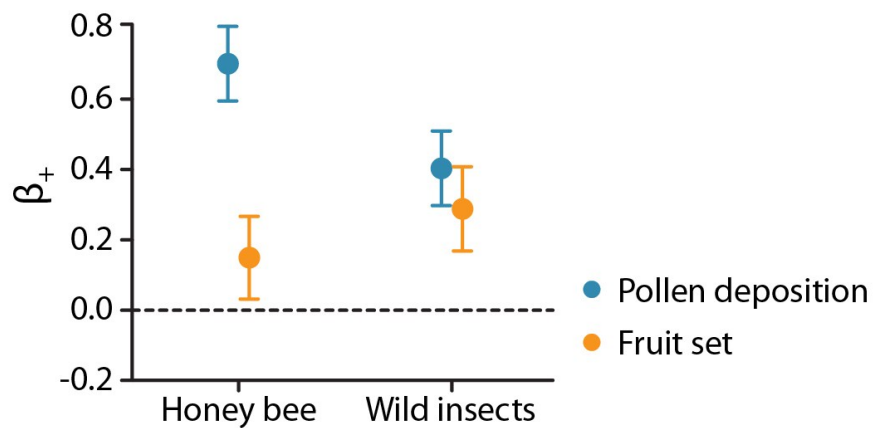
280 Compared to the large number of wild insect pollinators, pollinator research on the
281 honeybee has received relatively high attention, with particular emphasis being placed on the
282 current threats posed by parasites and disease (Brown, 2015; Genersch et al., 2010; Ratnieks
283 and Carreck, 2010). Whilst there is little doubt that honeybee losses are of concern, the

284 importance of all remaining wild pollinators and the threats they face should not be
285 overlooked (Figure 3; Ollerton et al., 2012; Winfree et al., 2007). Indeed, to have a
286 sustainable pollination service we cannot rely on one or a few domesticated species. Rather,
287 we must maintain a community constituting a diverse set of mobile pollinating organisms
288 with a broad suite of functional traits to maintain pollinators with ‘insurance values’ to meet
289 multiple pollination demands and prevent future pollination deficits (Albrecht et al., 2012;
290 Breeze et al., 2014; Gagic et al., 2015; Garratt et al., 2014b; Hoehn et al., 2008; Isbell et al.,
291 2011; Mallinger and Gratton, 2015; Martins et al., 2015; Orford et al., 2015; Winfree et al.,
292 2007; but see: Winfree et al., 2015). Crop pollination services can be enhanced when having
293 a diverse community of insect pollinators (Brittain et al., 2013a, 2013b; Greenleaf and
294 Kremen, 2006; Hoehn et al., 2008). Moreover, native and/or wild pollinators are more
295 efficient at pollinating certain plant species than honeybees (Figure 3; Garibaldi et al., 2014;
296 Jauker et al., 2012; Rader et al., 2013; Stanley et al., 2013; Thomson and Goodell, 2001;
297 Vicens and Bosch, 2000; Woodcock et al., 2013), and that long tongued wild pollinators can
298 effectively pollinate plants that may not be accessible for short-tongued pollinators such as
299 honeybees (Willmer, 2011). Yet despite the clear importance of wild pollinators, we have as
300 yet: i) a limited understanding of their population status; ii) major gaps in our knowledge of
301 the factors that threaten them and the subsequent impairment to the services they provide; and
302 iii) still much to learn about how we can change our actions and refine our approaches to help
303 maintain healthy pollinator populations and sustain valuable pollinator services.

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309 **Figure 3** | Figure taken and adapted from Garibaldi and colleagues (2013) who performed an
 310 analysis to investigate the pollination service of honeybees versus wild insect pollinators in
 311 41 animal pollinated crop systems across the globe. Plot shows the difference between the
 312 average regression coefficients (β_+ , $\pm 95\%$ CI) for pollen deposition and fruit set. The figure
 313 shows that whilst foraging honeybees deposit a higher amount of pollen on con-specific
 314 flowers compared to wild insect pollinators, they are less effective at pollinating the plant as
 315 measured by the level of fruit set which was higher in wild insect pollinators. This supports
 316 the importance of conserving a healthy community of native pollinators in these agricultural
 317 areas.

318

319 This paper posits that to sustain pollinator services and meet increasing pollination
 320 demands (Aizen et al., 2008a), we must understand how to support and maintain healthy
 321 pollinator populations and communities. The aim of this paper, therefore, is to provide a
 322 perspective on the research advances required for us to better understand the risks posed to
 323 wild pollinators and the services they provide, whilst detailing approaches we could adopt to
 324 better mitigate such threats. We touch on most of the factors previously implicated as threats
 325 to wild insect pollinators (see section 2) and consider the implications of our arguments to
 326 relate to the status of insect pollinators in agricultural and urban landscapes. We further place

327 much of our discussion in the context of the developed world where rural settings are often
328 dominated by intensive farming, and urban areas have well-developed infrastructures. Whilst
329 we also reference the global literature for our discussion of current knowledge and evidence
330 gaps, we concentrate on the schemes and initiatives taking place within Europe, and
331 especially the UK, for our case-studies to support future action plans and approaches.
332 However, the concepts, knowledge and perspectives we discuss here have broad implications
333 for the conservation of pollinators and their services worldwide.

334

335

336 **2. Major threats to the pollination service provided by insects**

337

338 Multiple factors likely act in combination to impose an overall level of stress on insect
339 pollinator populations (Bryden et al., 2013; Goulson et al., 2015; Perry et al., 2015;
340 Vanbergen et al., 2013). Implicated factors include climate change, habitat fragmentation,
341 land-use change and associated chemical inputs to the environment (i.e. agrochemicals),
342 invasive species and the added pressures of predation and competition (Godfray et al., 2014;
343 Gonzalez-Varo et al., 2013; Goulson et al., 2015; Kerr et al., 2015; Morales et al., 2013; Potts
344 et al., 2010a; Vanbergen et al., 2013, 2014a). Infection by parasites (and pathogens) also
345 poses a particular biotic threat, for example wild populations of the western honey bee, *Apis*
346 *mellifera*, have largely disappeared across Europe and N. America which may be attributed to
347 the parasitic mite *Varroa destructor* (Rosenkranz et al., 2010) and the viruses it transmits (Le
348 Conte et al., 2010; Sumpter and Martin, 2004). Managed honey bee colonies survive due to
349 significant prophylactic and reactive management techniques, and wild bumblebees are also
350 threatened by a range of emerging diseases (see section 4.3; N. America: Cameron et al.,
351 2011; UK: Fuerst et al., 2014; McMahon et al., 2015; S. America: Arbetman et al., 2013;

352 Schmid-Hempel et al., 2014). Given that many parasites appear to be both generalists and
353 transmitted via flowers, it is likely that solitary bees and other insect pollinators are also at
354 risk, but we currently lack the appropriate data to gauge such threats.

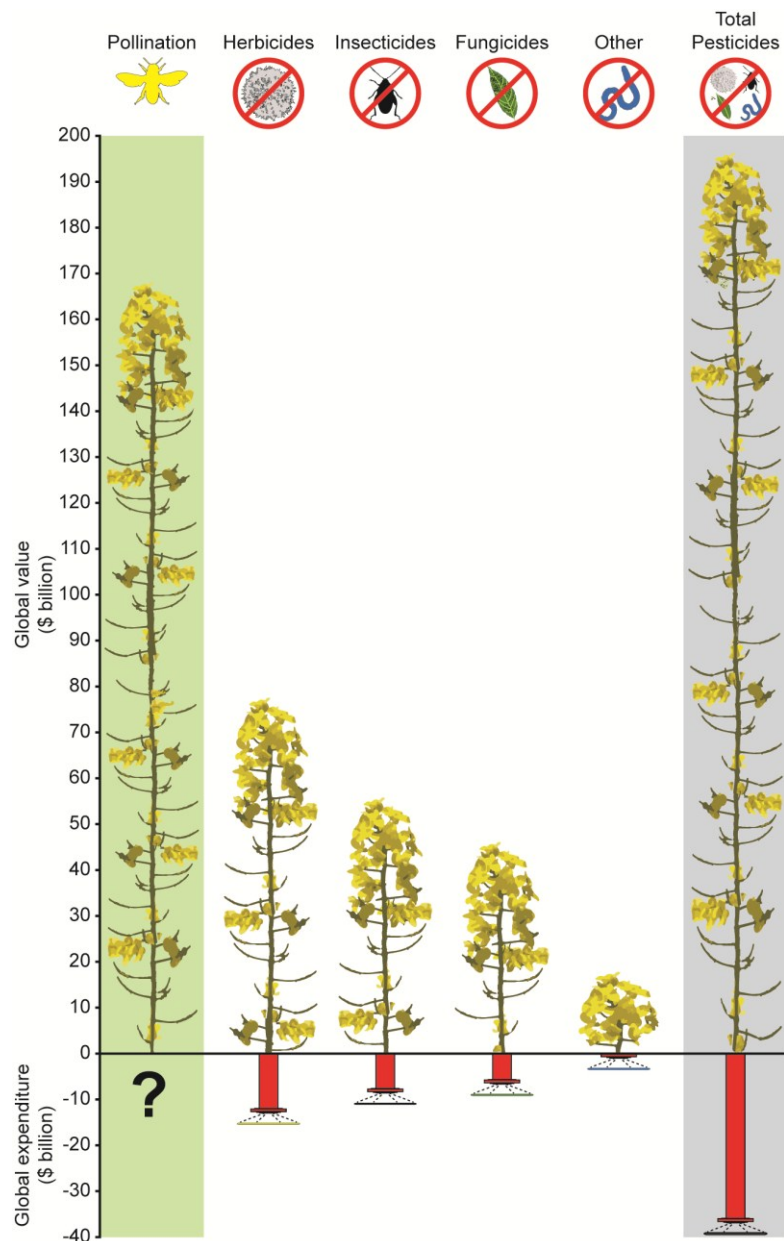
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356 Whilst efforts have been made to quantify the effect of each factor alone, gaining
357 reliable measures of their overall impact, particularly interactions, has proven difficult
358 because of large differences in morphology and life-history among taxa. Theoretical studies
359 suggest that a combination of stressors may be sufficient to trigger failure of social bee
360 colonies (Bryden et al., 2013; Perry et al., 2015), yet empirical studies looking at interactive
361 effects are typically limited to two (or few) factors (Baron et al., 2014; Becher et al., 2013;
362 Doublet et al., 2015; Fauser-Misslin et al., 2014; Gill et al., 2012; Gonzalez-Varo et al., 2013;
363 Hoover et al., 2012; Kennedy et al., 2013; Kleijn and van Langevelde, 2006; Oliver et al.,
364 2012; Pettis et al., 2013; Schweiger et al., 2010; Vanbergen et al., 2013). Land-use change
365 and management is seen as one of the leading drivers of insect pollinator declines (Garibaldi
366 et al., 2014; Ollerton et al., 2014; Vanbergen, 2014). The rapid habitat transformation over
367 recent decades, predominantly due to anthropogenic activity, appears correlated with reported
368 declines in insect pollinator species richness (Potts et al., 2010a; Senapathi et al., 2015).
369 Taking the UK as an example, we have seen at least a 97% loss of wild flower meadows
370 since the 1930s (Fuller, 1987) at the same time as a significant increase in the levels of
371 agricultural intensification and urbanisation (National Ecosystem Assessment 2011,
372 www.gov.uk/ecosystems-services). During this period, the UK has seen contractions in the
373 geographic ranges of a number of insect pollinator species, as well as declines in species
374 richness at local scales and species extinctions at the national scale (Biesmeijer et al., 2006;
375 Carvalheiro et al., 2013; Goulson et al., 2005; Nieto et al., 2014; Ollerton et al., 2014;
376 Senapathi et al., 2015; Thomas et al., 1994, 2004), although the rate of these declines may

377 have decreased more recently (Carvalheiro et al., 2013).

378

379 With increasing demands for food security and consumer pressure for affordable
380 produce, extensive areas of previously natural landscapes have been transformed to intensive
381 crop monocultures providing little nutritional resource to support insect pollinators if non-
382 flowering, or little nutritional diversity and/or quality if it is (Bates et al., 2011; Decourtye et
383 al., 2010; Garibaldi et al., 2011b; Jha and Kremen, 2013a; Kennedy et al., 2013; Kremen et
384 al., 2002; Raine and Gill, 2015; Steffan-Dewenter and Westphal, 2008; Tschardt et al.,
385 2005; Winfree et al., 2011a). Accelerating agricultural intensification across the globe over
386 the past five decades is deemed a major cause of biodiversity loss (Batary et al., 2011; Foley
387 et al., 2011). Land clearance and removal of hedgerows for intensive agriculture decreases
388 overall wildflower diversity and abundance (Carvell et al., 2006; Holzschuh et al., 2008;
389 Rundlöf et al., 2008) and can restrict suitable nesting sites (Goulson et al., 2010; Knight et al.,
390 2009; Williams et al., 2010). To protect crops against pest species, agrochemicals such as
391 pesticides are applied (Figure 4; Meehan et al., 2011), which may be having inadvertent
392 detrimental effects on insect pollinator behaviour, physiology, brood rearing and foraging
393 performance (Desneux et al., 2007; Exley et al., 2015; Fischer et al., 2014; Gill and Raine,
394 2014; Gill et al., 2012; Palmer et al., 2013; Whitehorn et al., 2012), leading to potential harm
395 of wild pollinator populations (Brittain et al., 2010; Rundlöf et al., 2015). This highlights one
396 of the ironies of modern agricultural practice; that by intensively growing flowering crops we
397 degrade the natural landscape that supports healthy pollinator populations and subsequently
398 degrading the very pollination service that the flowering crops benefit from (Figure 4).
399 Consequently, stress induced impairment to pollinator foraging ability could be having
400 negative impacts on the efficacy of the pollination service they provide (Gill and Raine,
401 2014), however to our knowledge, this expectation has not been explicitly tested. Viable



402 **Figure 4** | Relative value (financial income) and financial expenditure of pollination services
 403 and various pesticides. Value of pollination (green bar) was estimated to be worth \$168.75bn
 404 in 2009 (re-valuated from €153bn estimation by Gallai et al., 2009). Cost of these services,
 405 considered to be money invested into action to protect and promote pollinators through land-
 406 use management and health related interventions, however, is unknown. The cost of various
 407 pesticides was extracted from the United States Environmental Protection Agency for 2006-
 408 2007 Market Estimates. The value of each of these groups was then crudely calculated as an
 409 upper estimate of 500% return (Pimentel et al., 1993). The total value and expenditures of the
 410 four pesticides categories were summed for "Total Pesticides" (grey bar). "Other" pesticides
 411 include the nematocides, molluscicides and alternative chemicals (e.g. sulphur).

412

413 options to practice lower intensity farming and / or maintaining larger areas of semi-natural
414 habitats is likely to provide a more sustainable option in supporting insects for their
415 pollination service (see Section 4.6; Andersson et al., 2012; Bartomeus et al., 2014; Bohan et
416 al., 2013; Garibaldi et al., 2011b; Holzschuh et al., 2008; Kennedy et al., 2013; Kleijn and
417 van Langevelde, 2006; Ricketts et al., 2008).

418

419 For most insect pollinator groups, species richness has been reported to be in decline
420 across much of Europe (Biesmeijer et al., 2006; Bommarco et al., 2012a; Carvalheiro et al.,
421 2013; Conrad et al., 2006; Nilsson et al., 2013; Ollerton et al., 2014; Senapathi et al., 2015;
422 Wenzel et al., 2006) and N. America (Burkle et al., 2013; Cameron et al., 2011) over the last
423 several decades. Other areas of the globe may have also suffered from similar declines over
424 this time scale, but to date we have limited data to investigate this with any certainty. Our
425 understanding of major changes in insect pollinator species richness at broad spatial scales is
426 improving, yet we are still limited in what we can conclude from the data due to it being
427 either: i) at a low spatial resolution (coverage across large regional scales is patchy); ii) of
428 variable accuracy of location and sampling area (i.e. range from ± 100 s to $\pm 10,000$ s of
429 meters), which is problematic given that pollinators vary in dispersal ability; iii) limited to a
430 number of repeated measures at standardised times both within and across seasons / years; iv)
431 uses non-standardised sampling methods, making data comparison from multiple sources
432 difficult; v) obtained from multiple, different sources, which can introduce methodological
433 artefacts; vi) inconsistent in the level of taxonomic identification (i.e. down to family, genus
434 or species level); and, vii) perhaps most crucially, a general lack of data on species
435 abundance (at given spatial and temporal scales).

436

437 Identifying the drivers that shape insect pollinator population abundance requires both

438 high resolution data and universally standardised methods (Maes et al., 2013). The resulting
439 datasets should also be complemented by research focused on more localised population
440 changes. Whilst this type of data is missing for most insect pollinator groups, we can learn a
441 lot from data collected for butterflies, from databases such as for the UK Butterfly
442 Monitoring Scheme in which long term population data can allow powerful standardised
443 estimation of abundance in order to explore population trends and impacts of environmental
444 change (Oliver et al., 2010; Thomas, 2005; Thomas et al., 2011). Encouragingly this kind of
445 approach, further incorporating ‘citizen scientists’, is being adopted more widely, for
446 example by the National Pollinator Strategy for England (Figure 5) and the France-wide
447 monitoring scheme Spipoll (<http://www.spipoll.org>; Deguines et al., 2012). Gaining a greater
448 coverage of pollinator abundances, and thus pinpointing of localised population changes in
449 multiple locations across different geographic regions, enables a more coherent
450 understanding of the likely drivers of population change and consequent effects on ecosystem
451 services.

452

453

454 **3. Steps in the right direction to protect insect pollinator services: policy** 455 **actions**

456

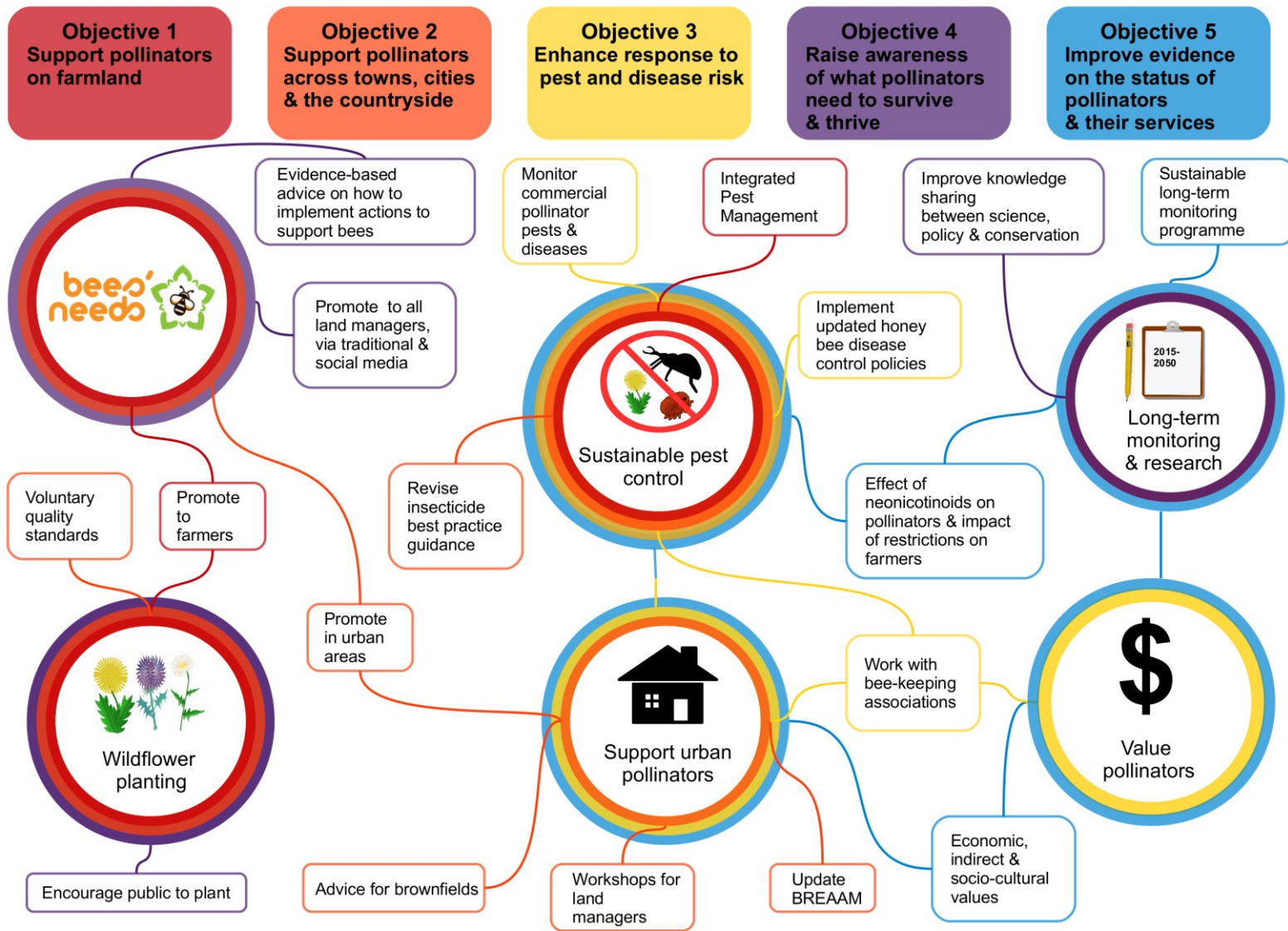
457 Mitigating threats to insect pollinator services has become an important issue for
458 policy and the public. A range of conservation initiatives have been developed to improve
459 understanding of the risks posed, and how to deliver a sustainable pollination service at
460 global, regional and national scales (Table 2). Advancing ecological research into insect
461 pollinators requires an understanding of where current policy stands and how research
462 approaches can answer or inform stated objectives. At the global stage, the Intergovernmental

463 Platform on Biodiversity and Ecosystem Services (IPBES; est.2012 and with 124 member
464 states), recognises the need to provide policy-relevant knowledge to inform decision making.
465 Identified as a globally important thematic topic, the assessment of “*Pollinators, pollination*
466 *and food production*” (incl. insect pollinators) is being conducted on the basis that
467 ‘pollination services’ are of fundamental significance. National pollinator initiatives /
468 schemes require robust evidence if stated objectives are to be well justified, achievable,
469 transparent, and critically to be of high societal relevance (see Table 2 in Appendix).

470

471 The recently launched National Pollinator Strategy for England (Defra, 2014)
472 provides an exemplary case study, in which an independent review of the “status and value of
473 pollinators and pollination services” was commissioned to provide a framework for future
474 action (Vanbergen et al., 2014a). This led to the declaration of four main objectives to: i)
475 support pollinators on a variety of land types; ii) enhance responses to pest and disease risk;
476 iii) raise awareness of what pollinators need to survive and thrive; and, iv) improve the
477 evidence on the status of pollinators and the service they provide. These objectives were
478 accompanied by statements of approach for how they should be achieved, by: a) developing
479 and field testing a new monitoring framework to provide abundance data for insect
480 pollinators and to integrate citizen science; b) assessing economic, social and cultural values
481 of pollinators; and, c) to better understand how agricultural practices, in particular crop
482 protection, affect insect pollinator populations (see Section 4 and Figure 5). The strategy also
483 encourages the research community to carry out longer-term studies that policy-makers need,
484 and policy makers to maintain a continuity of policy approach across different parliaments,
485 negating the traditional short-term government view that can often hinder conservation
486 actions (Pullin et al., 2009).

487



489 **Figure 5**| Recently launched National Pollinator Strategy for England (Defra 2014)
490 highlights five objectives to maintain sustainable pollinator services. Although each objective
491 addresses a particular key area, and has actions targeted to that area, many of the issues
492 addressed in the Strategy are interlinked, and each objective supports the other. Addressing
493 one action (open boxes) will in many cases contribute to another either directly or through
494 one of six main themes. Here, the actions of the Strategy are shown colour coded and
495 connected to allow navigation and an intuitive overview. Coloured rings show which
496 objectives are linked to a particular theme.

497

498 To support wild insect pollinators, agricultural land can be managed to deliver a broad
499 array of public goods, by combining food production alongside other ecosystem services such
500 as maintaining culturally valuable insect pollinator habitats. Within the EU, for instance, the
501 Common Agricultural Policy (CAP) can compensate agricultural land managers (i.e. farmers)
502 for direct losses to income when altering the intensity of farming practices to better support
503 biodiversity and public goods. Through agri-environment schemes (AES) financial
504 compensation is provided to help offset loss of income from setting-aside land that could
505 otherwise be cultivated ('opportunity cost'), and estimated implementation costs such as the
506 purchase of flowering plant-seed or costs of additional management (see sections 4.5 & 4.6;
507 Batary et al., 2011; Garibaldi et al., 2014; Rundlöf et al., 2013). AES can be used to target
508 ecosystem services that generate yield benefits in agriculture, such as insect pollination,
509 although this objective should be clearly separated from their use to conserve biodiversity,
510 because the necessary actions are likely to be different (Ekroos et al., 2014, Kleijn et al.,
511 2015). In England, the new AES launched in 2015, named Countryside Stewardship, includes
512 the 'Wild Pollinator and Farm Wildlife Package' designed to incentivise farmers to provide
513 additional resources for insect pollinators.

514

515 Pesticides used for crop protection pose a potential risk to non-target beneficial

516 insects, with foragers coming into contact with residues in the environment (Desneux et al.,
517 2007; Osborne, 2012). Particular concerns have been raised about the application of plant-
518 systemic neonicotinoid insecticides to flowering crops attractive to insect pollinators
519 (primarily bees). Reactions to the viewpoint that neonicotinoids place insect pollinators at an
520 unacceptable risk, the EU placing a restriction on the use of three neonicotinoids as seed
521 treatments (EFSA Commission Implementing Regulation (EU) no. 485/2013), and moves
522 haven been made in some states of Canada and the United States to reduce neonicotinoid use.
523 However, whilst restrictions may be viewed by some as a step-in-the-right-direction in
524 protecting insect pollinators, it has also been the subject of criticism, with questions being
525 raised over the assessment of the evidence underpinning this decision (Eisenstein, 2015) by
526 intensively growing flowering crops we degrade the natural landscape that supports healthy
527 pollinator populations and subsequently degrading the very pollination service that the
528 flowering crops benefit from. Whichever standpoint is taken, understanding the balance
529 between the benefit of crop protection and costs of impairment to a pollination service should
530 be the overall objective to achieve a pragmatic solution (Figure 4). Any review should
531 consider what the alternative methods of crop protection would be in order to sustain
532 adequate crop yields, to minimise off-site contamination to organisms other than insect
533 pollinators, and to be of low hazard to humans (Godfray et al., 2014; Raine and Gill, 2015).

534

535 **4. Understanding and mitigating specific threats to wild insect pollinators** 536 **to protect pollinator services**

537

538 Understanding the responses of insect pollinators to environmental stressors is
539 important in understanding how we can manage a sustained pollinator service for the future,
540 making it a major priority for both research and policy (Vanbergen et al., 2014a). Given the

541 complexity of these biological systems, we must avoid approaching pollinator management in
542 a naïve or overly simplistic manner otherwise our chances of meeting targets for pollinator
543 services may be compromised. For example, engineering areas of land set-a-side to support
544 pollinators cannot be assumed to be effective if landscape context and land-use type have not
545 been considered. Whilst the objective is to increase biodiversity, many actions will only
546 support increased numbers of common pollinator species rather than re-establishing
547 endangered species (Scheper et al., 2013). It would be unfortunate if advisory actions to
548 support pollinators were communicated to end users based on poorly informed decisions
549 resulting in little practical and conservation impact, as the ramifications are likely to result in
550 a loss of confidence in policy advice along with unwarranted financial costs (Pe'er et al.,
551 2014).

552

553 Key questions include how should we structure and maintain habitats to support insect
554 pollinators and increase subsequent services, and at what scale should this be implemented if
555 stakeholders are going to benefit, whilst ensuring that subsidies are not wasted. Here we
556 highlight some key evidence gaps, to refine our research efforts and ensure we better
557 understand how to maintain stable insect pollinator populations and structured communities
558 that are resilient to imposed stress. We consider seven primary themes, and for each we
559 discuss the evidence gaps and the research steps we can take to advance our understanding.
560 The steps raised within each theme are not in competition with each other, and in many cases
561 represent complimentary approaches in providing a generalised framework to direct future
562 research in addressing the evidence gaps..

563

564 **4.1 Understanding the stability of insect pollinator communities**

565

566 The stability of ecosystem functioning requires ecological resilience, which is the
567 capacity of a population, community or ecosystem to buffer environmental perturbations and
568 re-organise whilst undergoing change without loss of structure or functioning (Cardinale et
569 al., 2012; Folke et al., 2004; Holling, 1973). The key question is what level of perturbation
570 can be buffered when extreme events occur, such as an acute local pollution episode, severe
571 habitat disturbance or rapid climate change? Plant-pollinator mutualisms are embedded
572 within a wider network of community interactions (Memmott, 1999; Poccock et al., 2012).
573 While plant-pollinator interactions are dynamic, the topology or structure of the network is
574 thought to confer a degree of stability or robustness upon the community (Bascompte et al.,
575 2003; Kaiser-Bunbury et al., 2010; Memmott et al., 2004; Olesen et al., 2007; Ramos-
576 Jiliberto et al., 2012; Thebault and Fontaine, 2010; Tylianakis et al., 2010; Valdovinos et al.,
577 2013; Vieira and Almeida-Neto, 2015). There is, however, much debate over the extent that
578 different properties of network architecture confer stability or robustness to species loss
579 (Rohr et al., 2014). For example, species that are highly abundant and well connected to
580 many other species in the network, typically generalists, may increase overall network
581 robustness (Winfree et al., 2014). However, other evidence suggests that increased network
582 connectance (the proportion of possible links between species) may lead to a greater chance
583 of extinction cascades following species loss (Vieira et al., 2015), and the structuring of these
584 connections appears important with predictions that increased network nestedness imparts
585 stability (Allesina & Tang 2012; James et al., 2012). Moreover, behavioural plasticity means
586 that a pollinator species can potentially ‘rewire’ the network by switching to alternate plant
587 species following extirpation of a partner pollinator species, thereby maintaining the overall
588 community cohesion (Figure 2; Kaiser-Bunbury et al., 2010; Ramos-Jiliberto et al., 2012;
589 Valdovinos et al., 2013). Hence, this highlights the importance of ensuring we have a diverse
590 set of insect pollinators to maintain taxonomic or functional trait diversity (i.e. species

591 complementarity or functional redundancy), which serves to deliver a diverse set of
592 pollination services even in the face of species losses (Bartomeus et al., 2013; Moretti et al.,
593 2009; Woodcock et al., 2014; also see Kleijn et al. 2015).

594

595 Despite this apparent potential for an intrinsic stability to pollinator-plant networks,
596 there is increasing evidence of parallel declines in species richness of flowering plants and
597 their pollinators (Biesmeijer et al., 2006; Burkle et al., 2013; Cameron et al., 2011;
598 Carvalheiro et al., 2013; Ollerton et al., 2014). These declines may be linked to ecological
599 traits like diet specialisation or particular habitat requirements that predispose particular
600 pollinator species to extinction risk under environmental change (Aizen et al., 2012;
601 Biesmeijer et al., 2006; Williams et al., 2010). For example, in areas experiencing significant
602 transitions to more homogenous landscapes such as intensive arable monocultures, short
603 tongued generalist species of insect pollinators may be less affected than long tongued
604 specialists due to the concomitant loss of habitat(s) that maintain specific plants (Figure 2;
605 Goulson et al., 2005). Simulation modelling of empirical data shows that if the most
606 connected species in plant-pollinator networks are lost, then this may trigger a cascade of
607 secondary extinctions of plants or pollinators (Kaiser-Bunbury et al., 2010; Memmott et al.,
608 2004; Vieira and Almeida-Neto, 2015). The implication is that if environmental change
609 reaches a level sufficient to extirpate these highly linked species then there is a risk that the
610 whole plant-pollinator network could disassemble and jeopardise the continued delivery of
611 insect pollination in that ecosystem (Lever et al., 2014). Other research shows these highly
612 connected species tend to also be the most abundant and hence among the least vulnerable to
613 extinction under environmental change (Winfree et al., 2014). Recent advances in such co-
614 extinction modelling that incorporates variable dependences of different mutualistic partners,
615 and hence greater biological realism, should improve our ability to predict the likelihood of

616 extinction cascades and the role of network structure in community stability (Vieira and
617 Almeida-Neto, 2015).

618

619 There remain several notable research challenges. First, we need to better understand
620 how networks are assembled or disassemble under rapid environmental change or in response
621 to agricultural land management (Devoto et al., 2007; Vanbergen, 2014; Vanbergen et al.,
622 2014b). Second, we need to identify, through coupled experimental and predictive modelling
623 approaches the extent that different network properties (nestedness, connectance, modularity)
624 contribute to community dynamics or stability (Astegiano et al., 2015; Tylianakis et al.,
625 2010). Third, how can extinction-driven alteration of network structure affect the delivery of
626 crop pollination services and wild plant reproduction (Bohan et al., 2013; Tixier et al., 2013).
627 It is increasingly being recognised that species interactions, such as pollination, and the
628 services derived from them are an important measure of biodiversity that has been largely
629 overlooked due to the disproportionate emphasis on the species (nodes) within ecological
630 networks, rather than their links (Ings et al., 2009). For example, at what point does the
631 erosion or reorganisation of plant-pollinator networks by anthropogenic drivers lead to loss of
632 the links that confer functional redundancy upon the system and what is the magnitude of the
633 impact on pollination services and plant reproduction (Aizen et al., 2008b, 2012; Burkle et
634 al., 2013; Devaux et al., 2014; Stout, 2014; Vanbergen et al., 2014b)? To answer such
635 questions we need to explore the mechanistic linkages between network assembly and
636 pollination processes across replicated environmental gradients and experimentally
637 manipulate field systems. Coupled with this, we should consider how other scientific
638 disciplines have transformed our understanding and ability to quantify network structure and
639 function (Heleno et al., 2014). For example, pollinator network ecology is currently
640 underpinned by traditional field approaches (e.g. visitation transects) and taxonomic expertise

641 that provide data on the identity and frequency of mutualists but which are time-consuming.
642 Recent advances in molecular techniques has given rise to the possibilities of rapidly
643 analysing field collections to enable measures of taxonomic diversity and a high-throughput
644 quantification of species abundance (Bohmann et al., 2014; Keller et al., 2015; Richardson et
645 al., 2015), to re-construct with high precision and accuracy the structure of different plant-
646 pollinator assemblages (see next section 4.2; Vacher et al., 2015). These molecular
647 approaches will also enhance our ability to robustly monitor pollinator populations (Dicks et
648 al., 2013; Lebuhn et al., 2013), and quantify pollinator community structure under different
649 environmental stresses or contexts.

650

651 **4.2 Using molecular approaches to monitor insect pollinators**

652

653 Obtaining data to study insect pollinator population sizes and pollinator network
654 ecology are reliant on taxonomic identification of species and counts of individuals. For
655 monitoring purposes, quantitative surveys (recording richness and abundances of organisms)
656 are preferable to qualitative surveys (establishing only presence of species per locality: Kerr
657 et al., 2015), but both suffer from time consuming sorting, identifying and counting plant and
658 pollinator diversity, making it slow and financially costly for active management (Lebuhn et
659 al., 2013). Moreover, taxonomic experts are needed regularly but this may not be possible if
660 experts are few and in high demand. Researchers are therefore exploring the developments of
661 novel molecular tools to aid the efforts in both identification and quantification (Kuhlmann,
662 2015), with the aim of providing greater taxonomic precision, a robust method of monitoring
663 aspects of pollinator populations and to quantify pollinator community structure under
664 different environmental conditions (Tang et al., 2015). The overall objective is to monitor
665 reliably across multiple sites to gain spatial aspects of pollinator community structure

666 repeatedly over time to assess trends and react adaptively to protecting insect pollinator
667 services.

668

669 Employing molecular taxonomy through ‘DNA barcoding’ using the mitochondrial
670 cytochrome oxidase (COI) marker (particularly for insects) has now become relatively
671 commonplace (Hebert et al., 2003; Magnacca and Brown, 2012; Schmidt et al., 2015). By
672 cross-referencing sequences obtained from collected tissue samples to the growing global
673 database of barcodes that are linked to taxonomically well-studied specimens, we can use
674 these short, standardised gene regions to identify species rapidly and reliably (Figure 6).
675 Furthermore, such molecular identification is not limited by the state of the specimen, for
676 instance an insect pollinator specimen can be identified at any developmental life-stage or
677 even from non-living remains in the environment (eDNA) (Taberlet et al., 2012), or ingested
678 by predators. However, the frequently low differentiation between species and high
679 geographic variation within a species means we need to understand how well the COI marker
680 and its genetic variation capture the species limits. Thus, judicious use of COI-based genetic
681 clusters (sometimes referred to as Operational Taxonomic Units (OTUs)) is usually
682 acceptable for monitoring purposes, but these entities should be backed up by careful and
683 accurate morphological examination (Meyer and Paulay, 2005).

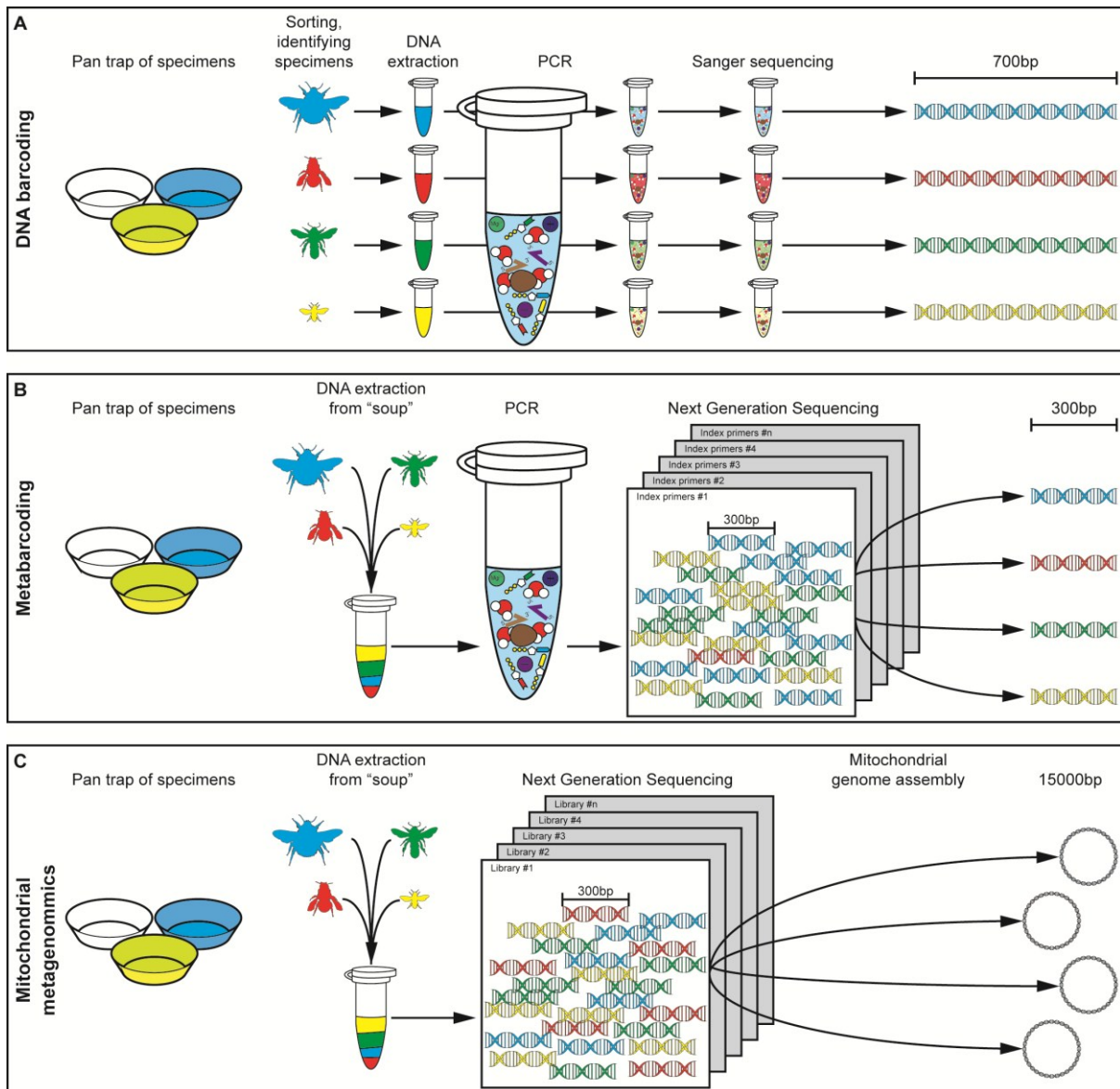
684

685 DNA barcoding of European bees has now progressed to the point where COI
686 sequences are available for many species (Schmidt et al., 2015) and are accessible through
687 the global Bee Barcode of Life Initiative (Bee-BOL) (<http://www.bee-bol.org>). Similarly,
688 there is a well-populated DNA barcode database for butterfly species (including all 59 British
689 spp.), but to date no orchestrated effort has been made to complete a DNA barcode database
690 for other pollinator groups. The practicality of populating these databases is aided by the

691 possibility to generate sequence data from preserved specimens, as those in museum
692 collections, which can establish the link to known vouchers of taxonomic relevance or to
693 monitor change over time against historical specimens (Hebert et al., 2013; Timmermans et
694 al., 2015). The resulting database of reference sequences then becomes part of a growing
695 framework of genotypes from wider surveys that extend the biogeographic, ecological and
696 taxonomic scope of pollinator studies.

697

698 Despite this power, DNA barcoding is time-consuming because it requires working
699 with individual specimens, and so next-generation sequencing (NGS) technology may be
700 preferred, which can potentially survey specimens in bulk and circumvent the need to
701 separate and sort samples (e.g. captured insect pollinators in a pan trap). The most
702 straightforward and cost efficient approach is ‘metabarcoding’ that applies the principle of
703 DNA barcoding to a bulk sample by conducting PCR and sequencing on the mixture allowing
704 an indiscriminate field collection to be analysed (Figure 6; Creer et al., 2010; Gibson et al.,
705 2014; Taberlet et al., 2012; Vacher et al., 2015; Yu et al., 2012). Massively parallel NGS
706 technologies produce numerous independent sequence reads, each corresponding to a
707 separate PCR amplicon, which may originate from any species in the mixture. In addition,
708 primer indexing permits the simultaneous sequencing of numerous samples (e.g. from
709 multiple pan traps), keeping costs low. However, this methodology is associated with various
710 problems such as biases in amplification success across different taxa, which may create false
711 negatives (Clarke et al., 2014; Tang et al., 2015), contamination risk and potential co-
712 amplification of mitochondrial pseudogenes (Song et al., 2008), and the comparatively short
713 sequence achievable with the current NGS technology, which limits the analysis of the COI
714 gene to roughly a half-length ‘minibarcoding’ and hence reduces discriminatory power (Tang et
715 al., 2015).



716

717 **Figure 6** | DNA barcoding, metabarcoding and mitochondrial metagenomics pipelines. **A)**
 718 Specimens collected (e.g. pan trap) must first be identified by expert taxonomists. For DNA
 719 barcoding, specimens and all downstream processes must be kept separate. The DNA from
 720 each representative specimen is extracted separately, amplified by PCR at specific loci (i.e.
 721 COI) and Sanger sequenced, to produce a reference database for use for metabarcoding (B)
 722 and mitochondrial metagenomics (C). **B)** Metabarcoding does not require the sorting of bulk
 723 samples. Whole samples are homogenised into a ‘soup’ and DNA is extracted directly from
 724 this. PCR is performed on this bulk DNA extract with primers designed for a shorter read
 725 length to accommodate the possible sequencing output of the next generation sequences.
 726 Multiple bulk samples can be analysed simultaneously if differently indexed primers are used
 727 for each sample; these indexed primers can be used to sort the samples out bioinformatically

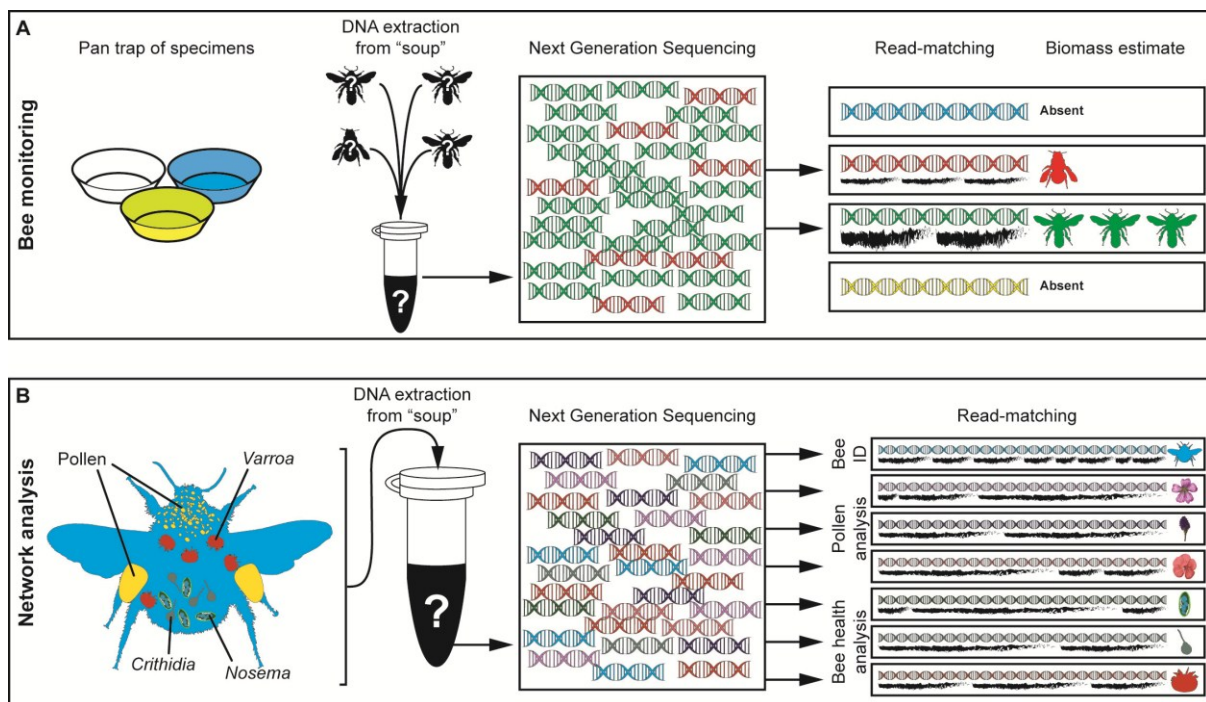
728 after sequencing. Next generation sequencing of the PCR products is performed after which
729 bioinformatics pipelines are used to extract unique sequence reads. These reads are
730 retrospectively identified using the DNA barcode reference database. C) Mitochondrial
731 metagenomics is also performed on bulk samples. DNA extracted from homogenised soups
732 do not undergo a PCR step, instead the raw DNA from the DNA extract is shotgun
733 sequenced. Different samples can be sequenced simultaneously if they are prepared in
734 differently labelled libraries. The sequence output is bioinformatically cleaned, quality
735 checked, parsed into mitochondrial DNA, and assembled into mitochondrial genomes. These
736 mitochondrial genomes are then identified using the reference database.

737

738 Given these drawbacks, the use of PCR-free methods is desirable. Current approaches
739 similarly analyse mitochondrial genomes, but exploit the fact that mitochondria are abundant
740 (ca. 200 copies per nuclear genome; Correa et al., 2012) and thus are naturally enriched over
741 other markers. Total DNA is extracted from a bulk sample of specimens and subjected to
742 shotgun sequencing to produce millions of short reads. Genome assembly from these reads
743 produce full or partial mitochondrial genomes for each of the species present in the mixed
744 sample. This approach, known as mitochondrial metagenomics (Crampton-Platt et al., 2015),
745 was recently applied to bee communities and provides powerful ‘super-barcodes’ used as a
746 reference library (Tang et al., 2015). These reference mitochondrial genomes are used to
747 profile any number and mixture of specimens that may be collected (for example in field pan
748 traps that attract and collect flying insect pollinators). Shotgun sequences from the bulk
749 sample are matched against the reference library thus revealing the presence and absence of
750 species from the read-matches. Crucially, this analysis requires many fewer reads than the
751 initial assembly of generating the mitochondrial genome references. Profiling success using
752 this approach has been high (>95%, taking into account false presences and absences
753 combined: Gómez-Rodríguez et al., 2015), and much higher than using metabarcoding (Tang
754 et al., 2015). In addition, studies have established the correlation of read number with
755 abundance and biomass (Gómez-Rodríguez et al., 2015; Tang et al., 2015; Zhou et al., 2013),

756 and thus read matching can potentially be used for assessment of species abundance (we
 757 therefore can obtain both qualitative and quantitative survey results; Figure 7). Tang et al.
 758 (2015) calculated that for large-scale monitoring of pollinator communities using the read
 759 mapping approach which is approximately 50% less labour intensive than the conventional
 760 survey work based on morphological identifications. Alternative approaches for abundance
 761 measures may come from rapid barcoding techniques conducted by PCR directly on the
 762 specimens (no DNA extraction) and short reads that could potentially sequence tens of
 763 thousands of specimens whose identification is based on sequence data on which to base the
 764 specimen count (Meier et al., 2015; Tang et al., 2015; Wong et al., 2014).

765



766

767 **Figure 7** | Read-matching of sequences. **A)** Specimens caught in a pan trap of unknown
 768 identity and quantity can be homogenised and shotgun sequenced. In this example, the only
 769 sequences present in the sample match to two of the four species in the DNA reference
 770 library. Furthermore, given that the number of sequences matching the amount of
 771 representative DNA in the extract, the number of sequence reads should correlate with the
 772 amount of DNA in the original sample, and given that the amount of DNA present per
 773 specimen should correlate with the biomass of the specimen, obtaining a quantification of the

774 number of specimens per species is possible with an appropriate calibration. **B)** Commensals
775 of the specimen can be elucidated with the read-matching approach. DNA from the carried
776 pollen, parasites and pathogens, as well as that from the pollinator, are present in the DNA
777 extract. In this example, the identity of the bee, the pollen from three plant species and the
778 disease agents are shown using a read-matching approach.

779

780 Beyond the estimates of abundance, the advantage of metagenomic sequencing is that
781 the sequence reads are mostly a complete reflection of the entire ecosystem, including those
782 organisms associated to the pollinator specimens (Bohmann et al., 2014). These methods
783 offer the potential to study plant-pollinator networks as DNA from bulk samples contains not
784 only pollinator DNA, but also any associated organic material such as pollen, parasites and
785 pathogens (Figure 7). Using a read-matching approach, matching DNA from pollen would
786 allow the practitioner to populate plant-pollinator networks in a more direct fashion than
787 studies using visitation as a proxy (King et al., 2013), and more rapidly, reliably and to a
788 higher resolution than microscopy methods (Kraaijeveld et al., 2015). Presently plant
789 identification via molecular approaches still rely on several short chloroplast barcodes but
790 whole chloroplast genome sequencing is expanding rapidly, which will overcome the
791 problem of low chloroplast abundance in the pollen, currently addressed with sequencing of
792 the nuclear ITS region (Bohmann et al., 2014; Keller et al., 2015; Richardson et al., 2015;
793 Sickel et al., 2015). Matching parasites and disease agents (section 4.3), and the associated
794 pollinator microbiome, would allow assessment of pollinator community health as well as its
795 diversity and abundance. Metagenomics could be used to detect EIDs in wild pollinators, a
796 largely unknown and neglected system or even the spill-over of these diseases from managed
797 to wild pollinators (Goulson and Hughes, 2015). While molecular approaches have been
798 tested to screen bees for disease agents (Fürst et al., 2014; Graystock et al., 2014), these are
799 PCR based and therefore target specific diseases with primers that need to have close affinity
800 with the targets. A metagenomics approach does not suffer from a primer biases and so it

801 would be possible to detect all of the diseases simultaneously.

802

803 **4.3 How do parasites shape wild insect pollinator populations?**

804

805 Wild pollinators exist in complex and diverse assemblages. In unperturbed
806 assemblages, endemic parasites may control populations of individual species when showing
807 host specialisation, or, if they are multi-host parasites act through apparent competition to
808 modulate the relative abundance of different species (Dobson, 2004; Holt and Pickering,
809 1985). In perturbed assemblages, where the abundance of subsets of species is depressed
810 through human impacts, effects of reservoirs (where one host acts as the major carrier of the
811 parasite) and spill-overs (where the parasite is transmitted from reservoir to non-reservoir
812 hosts) may drive host species with anthropogenically reduced populations to extinction
813 (Daszak 2000; de Castro and Bolker, 2005; Dobson 2004). Emerging infectious diseases,
814 which are not confined to pollinators, may also threaten population declines or species
815 extinction if they spread rapidly and have high impacts on individual fitness (de Castro and
816 Bolker, 2005). Hence, understanding the top-down pressure placed on insect pollinator
817 populations by endemic and emerging parasites, alongside infectious diseases (EIDs), is
818 important given that maintaining healthy populations is a prerequisite for a healthy
819 pollination service (Brown, 2015). For example, concerns have been raised over the effect
820 that parasites and infectious diseases are having on the overall pollination service provided by
821 honeybees, as individual infection can impair individual foraging performance (Wolf et al.,
822 2014), and high prevalence in colonies can lead to significant problems (Dainat et al., 2012;
823 Higes et al., 2008; Martin, 2001; Nazzi et al., 2012).

824

825 Our understanding of endemic parasites and EIDs in wild pollinators is largely limited

826 to bumblebees, although parasites and disease in managed honey bees are also well known
827 (Genersch et al., 2010). Our knowledge of the situation in wild solitary bees, hoverflies, and
828 other pollinators, is limited to point samples of individuals in space and time (Evison et al.,
829 2012; Singh et al., 2010). Whilst our knowledge of endemic parasites in bumblebees is
830 relatively broad and deep (Schmid-Hempel, 1998), we know almost nothing about how they
831 interact with the population dynamics of their hosts. This is a specific example of the larger
832 issue, in that we simply do not understand the relative importance of the putative drivers of
833 wild pollinator population dynamics in the field. Similarly, we have only a vague idea of the
834 distribution and abundance patterns of endemic parasites, although studies using regional
835 (Durrer and Schmid-Hempel, 1995), and continent-scale (Cordes et al., 2012) approaches are
836 starting to fill this gap. In contrast, studies of EIDs in wild bumblebees have explicitly taken
837 large-scale geographical approaches, with parasites, some associated with population
838 declines, being mapped in N. America (Cameron et al., 2011), S. America (Schmid-Hempel
839 et al., 2014) and the United Kingdom (Fürst et al., 2014; McMahon et al., 2015).

840

841 If we are to understand the impact of endemic parasites and EIDs on wild pollinator
842 populations and its subsequent effect on pollinator services, a number of key questions need
843 to be addressed. First, we need to map the drivers of population dynamics and identify the
844 relative importance of endemic parasites and EIDs within this larger picture. This requires
845 studies of population dynamics within and across years at the same site (see next section 4.4),
846 and the quantification of potential limiting resources, predators, and parasites. Second, the
847 directionality of EIDs needs to be confirmed – who is giving what to whom across the
848 season? And the degree of impact of EIDs on wild pollinators needs to be determined in
849 laboratory studies (Graystock et al., 2013). Third, we need to understand the drivers of
850 parasite population and epidemic dynamics within and across species. This will require a

851 range of approaches, including eco-immunology, field observation and landscape analyses.
852 Specifically, we should be attempting to map the ‘parasite exposure landscape’, that is, the
853 distribution and abundance of parasites in the environment (both in hosts and on flowers;
854 Anderson et al., 2013), and how this relates to the probability of becoming infected. This is
855 likely to vary over spatial and temporal scales across the landscape, but by identifying key
856 ‘hubs of transmission’ we may be able to focus our conservation efforts, and investigate
857 whether such hubs geographically overlay with hotspots of high pollination demand.

858

859 **4.4. Understanding insect pollinator population responses to resource availability**

860

861 Meeting local pollination demands reliably and throughout the year requires large
862 insect pollinator populations to provide numerical and functional responses to temporal
863 changes (Lautenbach et al., 2012; Polce et al., 2014), and to be stable with long-standing
864 residency in local areas (Klein et al., 2012). A realistic concept of a stable population is one
865 that shows resilience to environmental perturbation by rapidly recovering from induced
866 reductions in population size (a ‘bounce back’; Holling, 1973; Steiner et al., 2006). It is
867 therefore important we understand how populations respond to imposed stress to protect
868 populations from reaching tipping points that trigger population crashes (Bryden et al., 2013;
869 Dai et al., 2012; Drake and Griffen, 2010; Lever et al., 2014). Population resilience is largely
870 dependent on population size, with small populations being vulnerable to bottleneck events
871 with proportional losses of individuals, compared to large populations, likely having a higher
872 impact on effective population size (Fagan et al., 2001; Hanski and Saccheri, 2006; Shaffer,
873 1981), increasing population susceptibility to future stress and chances of inbreeding
874 depression (Frankham, 1995a, b) and ultimately local extinctions (Lawton and May, 1995;
875 Saccheri et al., 1998). We therefore need to understand what habitat requirements are

876 important for pollinator life-histories, and how variation in resource availability impacts on
877 population dynamics (Yamamura et al., 2006).

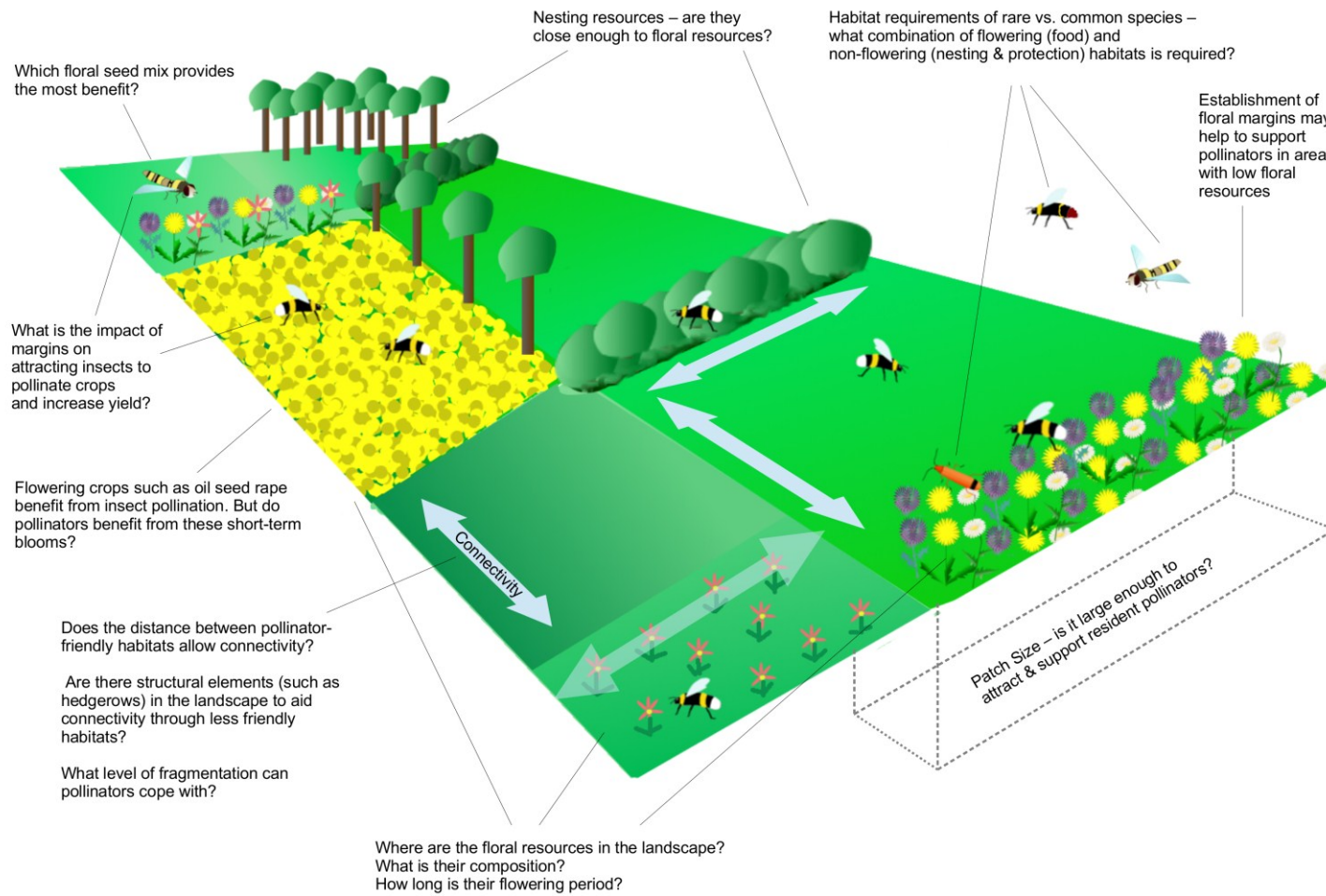
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879 We can start by first targeting research to help conserve key species or groups that
880 provide a highly valued pollinator service (Kleijn et al., 2015, Winfree et al., 2015). In
881 designing and managing supportive habitats, we need to understand a habitat's carrying
882 capacity for the pollinator(s) in question, taking into consideration habitat size (which can
883 influence resource diversity, potentially resource quality, and positively correlates with
884 absolute resource abundance although not necessarily resource density) and compositional
885 heterogeneity (Blaauw and Isaacs, 2014b; Fahrig et al., 2011; Tschardt et al., 2002).
886 Importantly, we must consider particular habitat features that meet fundamental life-history
887 requirements (Blaauw and Isaacs, 2014a, Dicks et al., 2015). For an insect pollinator, the
888 primary habitat requirements are likely to be: i) floral and other nutritional resources, ii) nest
889 (breeding) sites, and iii) levels of protection (Figure 8; Fortel et al., 2014; Williams and
890 Kremen, 2007; Wray and Elle, 2015). The floral preferences of a large diversity of insect
891 pollinators have been well studied (see section 4.6; Willmer, 2011), and significant advances
892 have been made in understanding how to attract species or communities to engineered or
893 enhanced floral rich habitats (although other predatory or parasitic insects require other
894 available organisms to survive). However, we understand little of whether such habitats are
895 able to 'retain' resident populations in the long term with any constancy. Therefore, shifting
896 some of the research focus towards better understanding requirements ii and iii, can help us to
897 understand the 'retention capability' of habitats. To do this we need to understand more about
898 the micro-habitat requirements for nesting (breeding) sites and where these sites are spatially
899 and temporally located. Though our knowledge base on wild bees is improving significantly
900 (Goulson et al., 2010; Knight et al., 2009; Williams et al., 2010), we rarely find nests that fail.

901 Furthermore, we understand even less about what composite features of a habitat provide
902 protection from threats such as predators and weather conditions (Boggs and Inouye, 2012).

903

904 Spatial and temporal variation of resources will have impacts on the basic properties
905 underpinning population dynamics, namely: i) the basic demography (birth and death rates)
906 and age structure of a population; and, ii) the immigration and emigration of individuals in
907 and out of local (sub-)populations. Although assessment of simple birth rates appears at first-
908 glance a basic task, it is in actuality rather challenging, as insect pollinators are often small
909 and hard to track (Hagler and Jackson, 2001), with nest sites often being inconspicuous.
910 Unlike honeybees where close monitoring of colonies has provided detailed insights into
911 social colony dynamics (Becher et al., 2013, 2014; Khoury et al., 2011, 2013; Naug, 2008),
912 there is a comparative lack of data on the levels of successful hatching, rearing and adult
913 development for most insect pollinators. Gaining knowledge on seasonal emergence times
914 and dispersal behaviour would inform us of the best times for sampling pollinators to provide
915 indirect data on hatching successes. Furthermore, development of methods to help find and
916 identify nests aided by local knowledge of pollinator natural history, would be advantageous.
917 Pollinator fecundity and hatching success is likely to be related to the availability of
918 nutritional resources (O'Brien et al., 2003; Vaudo et al., 2015), which may be increased by
919 providing enhanced floral resources in the landscape (Dicks et al., 2015; Holland et al., 2015;
920 Morandin and Winston, 2006). But to effectively support a population throughout the season
921 comprehensive knowledge of plant-pollinator phenologies are required to ensure that
922 resources are available for crucial stages of the life-cycle (Memmott et al., 2010), as biasing
923 resources towards specific age sector(s) of the population may leave other sectors vulnerable.



924

925 **Figure 8**| Representation of a rural scene, depicting a typical patchwork of habitat types across the landscape (not to scale). This highlights some
 926 of the factors that influence residency and movement of pollinators across the landscape.

927

928 Susceptibility to different environmental pressures is likely to vary at different life
929 cycle stages of an insect pollinator. Gaining a better understanding of this will enable
930 implementation of habitat features that can increase protection at different times of the year.
931 Floral resource availability, for example, is not only important leading up to laying, but also
932 crucial to build nutritional reserves in preparation for species that hibernate (i.e. storing fats)
933 if starvation is to be avoided (Beekman et al., 1998; Brown et al., 2003; Leather et al., 1993),
934 and flowers should be located relatively close geographically to hibernation sites. Moreover,
935 the overwintering sites need to be suitable to enable individuals to overcome the demands and
936 exigencies of hibernation. Hence, efforts should be refined to aid individuals at critical life
937 stages in order to mitigate populations experiencing severe ‘demographic pinch points’.

938

939 Repeated and long term monitoring of species at specified sites, including employing
940 mark-recapture methods, will be key to understanding the population biology of insect
941 pollinators (see sections 2, 3 and all sub-sections of section 4; Drag et al., 2011; Hagler and
942 Jackson, 2001). However, observation of individuals in a habitat patch may represent a
943 temporary immigration, instead of a permanent population enhancement (Jonsson et al.,
944 2015), a tactic used by pollinators to access requirements that may be limited in other
945 neighbouring patches – a process known as ‘habitat complementation’ (Dunning et al., 1992;
946 Mandelik et al., 2012; Tarrant et al., 2013). Hence, dispersal ability is likely to determine
947 how well such meta-populations can cope with increasing habitat fragmentation (Fahrig,
948 2001), an area that would benefit from further research (Jauker et al., 2009; Rands, 2014).
949 Studies have shown that foraging abilities of flying insect pollinators range widely (Cant et
950 al., 2005; Carvell et al., 2012; Gathmann and Tschamtker, 2002; Greenleaf et al., 2007;
951 Knight et al., 2005; Osborne et al., 1999; Pasquet et al., 2008), and this will be further

952 influenced by climatic variables, such as temperature and wind-speed. To inform habitat
953 management practices, efforts should be focused on advancing our understanding on the
954 foraging / dispersal abilities in different landscape complexities for a diverse set of
955 pollinators (Figure 9), taking into consideration the combination of size and eco-
956 physiological strategies (Gathmann and Tschardtke, 2002; Niitepold et al., 2009; Pawar et al.,
957 2012), and how such demands may influence susceptibility to other threats. In a rapidly
958 changing world, it is important to understand how meta-populations ranging from numerous
959 small (and fragmented) local populations showing low connectivity (low gene flow), through
960 to fewer large and well-connected populations (high gene flow) can cope under different
961 environmental settings and landscape structural complexity (Figure 8; Tschardtke et al.,
962 2002; Weibull et al., 2000), and crucially how this has a consequent effect on pollination
963 service (Holland and DeAngelis, 2001).

964

965

966 **4.5 Engineering flowering field margins as habitats to attract insect pollinators**

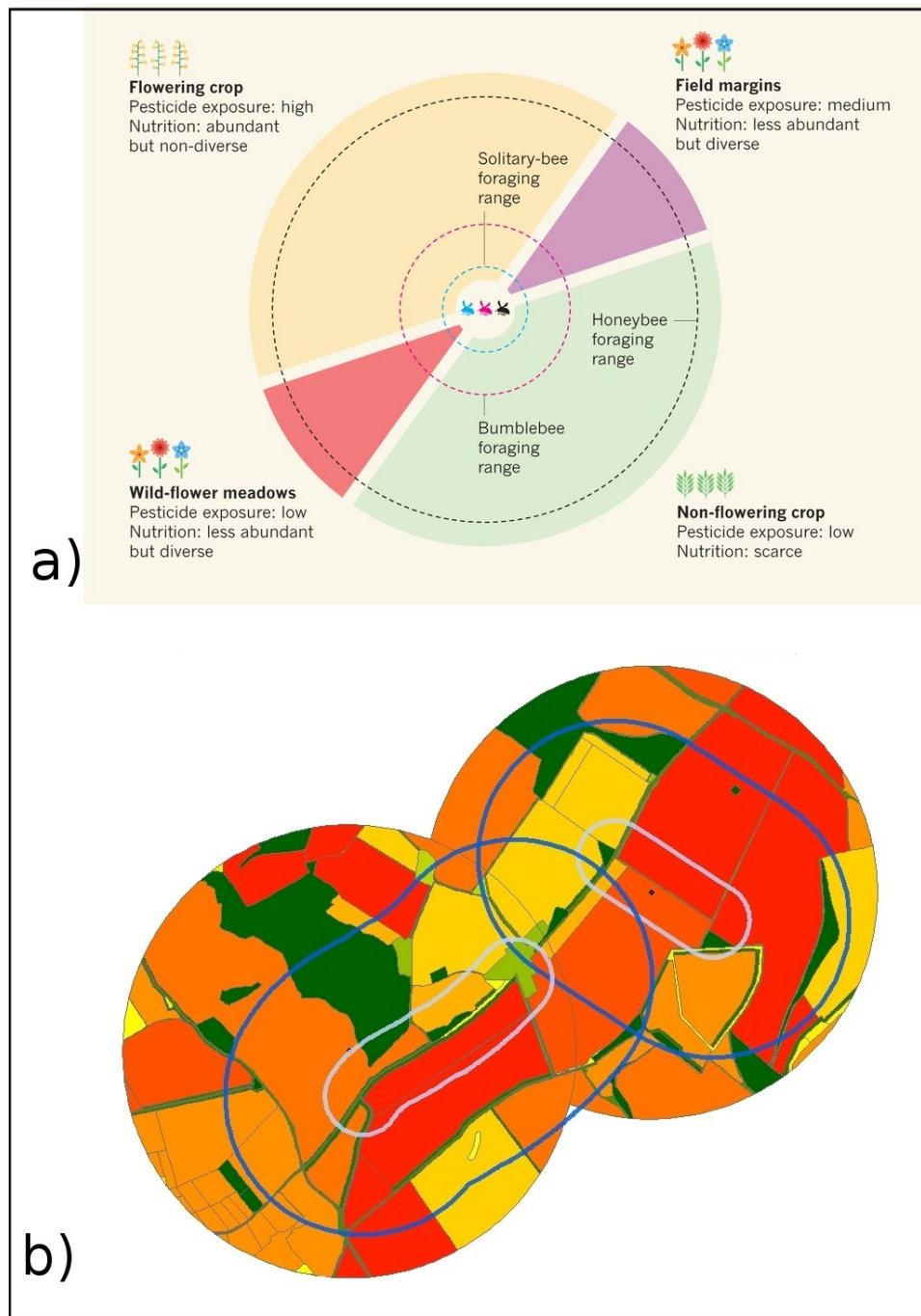
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968 Engineering of certain habitats can increase the delivery of floral resources, nesting
969 areas and protection to support a greater insect pollinator abundance and richness of insect
970 pollinators which can in turn provide a pollination service to improve crop yields, including
971 its reliability and quality (Klatt et al., 2014; Kleijn et al., 2006). Unfortunately, understanding
972 how to engineer the ‘optimal’ agricultural field margin has not proved to be simple to
973 achieve. Crops differ in the level of pollination and the community of insect pollinators
974 required (Albrecht et al., 2012; Fruend et al., 2013), and pollinator species may also differ in
975 their resource requirements throughout the year (Olesen et al., 2008). Wild insect pollinators
976 also differ in their mobility and foraging strategies (Greenleaf et al., 2007) and consequently

977 the amount and spatial configuration of engineered habitats should determine their
978 effectiveness (Brosi et al., 2008). Furthermore, engineered margins are not necessarily always
979 needed or desirable; a farmer growing predominantly non-flowering crops does not
980 necessarily need a pollination service, so if they were to invest in engineering a pollinator
981 supportive margin the purpose would be for ecological (rather than economic) reasons. The
982 requirement of engineered margins, therefore, needs to be determined with guidance on
983 regional or landscape specific pollinator community abundance or on-farm monitoring
984 schemes. Moreover, relying on engineered margins alone is risky because establishment of
985 the sown species can fail and this requires removing land from production. Therefore, a
986 better option may be to supplement existing habitats that also support insect pollinator
987 communities (Franzen and Nilsson, 2008; Ricketts et al., 2008; Wood et al., 2015a). Thus
988 when managing floral rich habitats we should consider whether it will compliment, rather
989 than conflict, with other blooming periods from other floral sources nearby (Williams et al.,
990 2010), and acknowledge that this will differ in areas where crops are non-flowering versus
991 flowering (Figure 9; Raine and Gill, 2015). The value of such natural or semi-natural habitats
992 in agricultural areas remains largely undefined although national and international projects
993 (www.queessa.eu) are underway to address this gap.

994

995 The first step in designing engineered margins is to identify whether a pollination
996 deficit exists for crops, and the extent to which this deficit varies between regions with
997 different landscape complexities (Marshall et al., 2006). Variation is expected because
998 existing habitats will differ in the floral resources they provide and in their abundance and
999 distribution across landscapes. The level of deficit is known for some crops (Aizen and
1000 Harder, 2009; Lautenbach et al., 2012; Polce et al., 2014), although varietal differences exist
1001 and are not well understood (Klein et al., 2007). Information about landscape-scale provision



1003

1004 **Figure 9** | Foraging options to central place foragers (such as bees) is dependent on their
 1005 foraging range. **A)** Taken from Raine & Gill (2015) this is a conceptual diagram showing
 1006 some general types, and typical proportions, of habitats that are available to foraging
 1007 pollinators in rural landscapes. Pollinator species show a wide range in dispersal abilities
 1008 which can limit the total amount of suitable habitats that can be viably visited. Evidence
 1009 shows that individual honeybees have a much larger foraging range than either bumblebees

1010 or solitary bees (see section 4.4). **B**) Foraging ranges of solitary bees drawn around the
1011 location of an engineered floral field margin strip (black line). Map depicts the real cropping
1012 and semi-natural habitats found on a farm in Dorset, UK, with 160m representing the
1013 minimum (grey line) foraging range and 600m the maximum (blue line) (Gathmann and
1014 Tscharntke, 2002). Ranges allow a visualisation of: i) which areas the field margins are likely
1015 to have an effect on, regarding attraction of resident solitary bees; ii) which areas bees are
1016 able to reach were they to nest or reside in one of the field margins and act as a central
1017 forager. Pollinators with small foraging ranges have the potential to be isolated from
1018 neighbouring populations; intriguingly even at a maximum foraging distance, solitary bees
1019 may be unable to travel between field margins and thus connecting habitats are likely
1020 required in such a setting (the landscape map was generated for the QuESSA project funded
1021 by the EU and provided with permission from John Holland).

1022

1023 is scarce (Richards, 2001; Steffan-Dewenter et al., 2002), but is needed if the requirement for
1024 additional floral or nesting resources is to be identified spatially and temporally (Figure 8).
1025 Engineered margins have typically been designed to support bumblebees by planting their
1026 most preferred foraging plants (Fabaceae), however, evidence is now emerging about the
1027 value of other insects that have different foraging preferences (Wood et al., 2015b;
1028 Woodcock et al., 2013). Data are limited but a few studies suggest that perennial wildflower
1029 mixes may support a wider range of pollinators (Dicks et al., 2010), and floral richness is as
1030 important as floral abundance, with insect pollinators shown to have increased growth rates
1031 and immuno-competence from being fed diverse diets (Di Pasquale et al., 2013; Vaudo et al.,
1032 2015). Achieving the desired habitat, however, does present challenges because sown plant
1033 species may not always establish and can depend on soil type, the plant's establishment
1034 method, competition between sown and unsown plants and subsequent management (Pywell
1035 et al., 2011).

1036

1037 We next need to understand how such engineered habitats are utilised by insect

1038 pollinators and whether they can realistically subsidise nearby crop pollination. The value of
1039 flower-rich habitats is typically measured through assessment of pollinator abundance and
1040 species richness relative to other habitats (Carvell et al., 2007), and is likely influenced by the
1041 composition and complexity of the landscape (Scheper et al., 2013; Shackelford et al., 2013).
1042 Surprisingly, it is still poorly understood whether the attraction of pollinators to flowering
1043 crops is enhanced with nearby engineered habitats, although the positive influence of semi-
1044 natural habitats has to some extent been demonstrated (Chacoff and Aizen, 2006; Kleijn and
1045 Sutherland, 2003; Klein et al., 2003; Kremen et al., 2004; Morandin and Winston, 2006).
1046 Even where the level of crop pollination has been quantified it is usually measured by proxies
1047 such as pollinator abundance in the field or resulting crop yield, rather than pollen transfer,
1048 visitation rates or plant fitness (King et al., 2013; Liss et al., 2013).

1049

1050 Quantifying the measures needed to achieve the desired pollinator community, is
1051 important, whether these are enhancement of existing habitats or establishment of new
1052 flower-rich and nesting areas. Guidelines on the proportion of flower-rich habitats needed in
1053 the landscape are appearing (Holland et al., 2015; Morandin and Winston, 2006), but also
1054 with evidence of redistribution rather than population enhancement that require further
1055 investigation (see section 4.4; Jönsson et al., 2015). To maximise the potential of engineered
1056 margins and minimise the amount of land removed from production to support pollinators
1057 (Figure 10), the following steps could be adopted. i) For each flowering crop and main
1058 varieties identify whether pollination is needed and which pollinators are likely to provide
1059 this service. ii) Assess the landscapes in which the crops are produced for alternative floral
1060 and nesting resources through the pollinator activity period to identify potential resource gaps
1061 (see section 4.4). iii) Engineer margins using plant species that will not encourage crop pests
1062 or diseases. iv) Carry out small-scale trials of engineered margins to compare different

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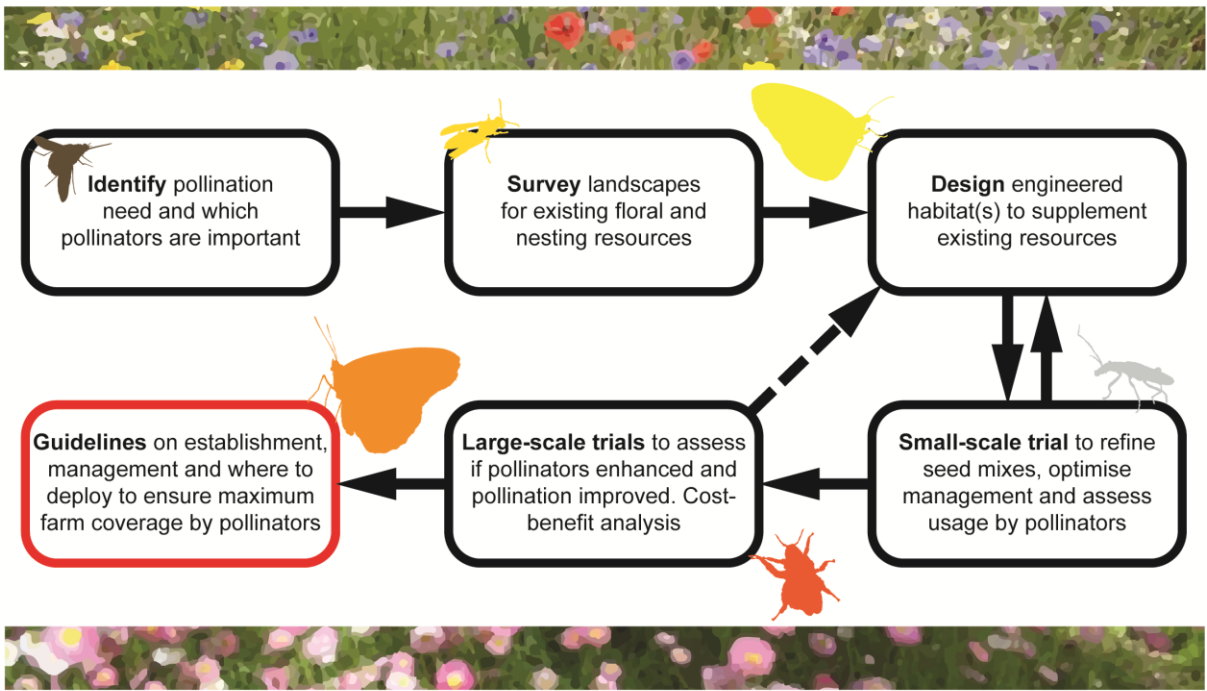
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Figure 10 Six primary steps to maximise the potential of engineered margins to support wild pollinators whilst minimising the amount of land removed from production. Between the Design and Small-scale-trial stages feedback will be required to ‘optimise’ management towards the target pollinator(s). Scaling up from Small-trial to Large-trials may have different and unforeseen outcomes, therefore, the dashed line represents feeding this information back to the design stage if this were to occur.

establishment methods across a range of soil and climatic conditions, and measure usage by pollinators and how this is enhanced within adjacent crops. v) Conduct larger-scale trials of engineered habitats across the landscape. vi) Evaluate the economic implications (a type of cost-benefit analysis) of removing land from production to provide the service in comparison to the increase in crop value. Step v should address spatial scales that consider abundance decay curves (estimated time of disappearance) from engineered habitats, requirements for corridors and existing resources provided by other non-crop and cropped habitats, and it should also evaluate whether crop pollination is improved relative to that in control areas.

1092 Such an approach could be conducted in conjunction with habitat development for natural
1093 enemies that also function as pollinators (e.g. certain hoverflies) or require similar resources
1094 (e.g. parasitoids that require nectar; Wratten et al., 2012).

1095

1096 **4.6 How might we improve the wider countryside to support insect pollinators**

1097

1098 A number of mitigation strategies are being implemented to conserve or enhance
1099 threatened biodiversity in agricultural landscapes. These often integrate conservation
1100 objectives with extensive farming practices in the same landscape, although many of these
1101 actions target more generic provision of resources for a range of wildlife rather than
1102 pollinators *per se* (Pywell et al., 2012). Specific actions of the AES for pollinators, such as
1103 retaining patches of uncropped flower-rich habitat and of planting floral resources (e.g. for
1104 declining bumblebee species), can increase the local abundance and species richness of
1105 foraging pollinators (Scheper et al., 2013; Wood et al., 2015b). In England, there is also now
1106 a Wild Pollinator and Farm Wildlife Package in the Countryside Stewardship that aims to
1107 support pollinators through provision of flower-rich and nesting habitats. A few studies show
1108 that such actions also enhance pollination services to nearby crops (see section 4.5; Garibaldi
1109 et al., 2014), but whether this reflects genuine increases in numbers or pollinators moving
1110 around with consequent dilution of densities elsewhere is largely unknown (Carvell et al.,
1111 2014). So far, there is very little evidence that these actions make pollinator communities
1112 more resilient, which the underlying policy aim. From a policy perspective, the key questions
1113 are: i) what area and extent of land should be dedicated to such actions; ii) how should they
1114 be deployed spatially to have the best chance of success; and, iii) how does it work in practice
1115 (Dicks et al., 2015).

1116

1117 To date, the main focus for pollinator conservation actions has been on the design and
1118 provision of mixtures of flowers, whilst largely ignoring other potentially limiting factors,
1119 especially in temperate regions. These actions have traditionally focused on providing
1120 particular taxa with specific floral needs: for example, the response of declining, longer-
1121 tongued bumblebees to the creation of legume-rich margins has received considerable
1122 attention (Heard et al., 2007). More recent studies have tried to tackle the question of how
1123 much flower-rich habitat is needed across different landscapes (Holland et al., 2015),
1124 including detailed studies of how bumblebees respond to flower provision (Carvell et al.,
1125 2012; Heard et al., 2007), or how they alter their foraging patterns (Carvell et al., 2014), by
1126 estimating how much pollen and nectar some common species need to raise their larvae
1127 (Dicks et al., 2015). This has direct relevance as to whether the encouraged minimum area of
1128 land set aside within the AES is sufficient, as evidence suggests that only common species
1129 may benefit from current guidelines (Holland et al., 2015; Wood et al., 2015b). Therefore, the
1130 question is whether we need to achieve some threshold percentage area before we will see
1131 any significant effects on supporting stable insect pollinator assemblages? However, without
1132 long-term pollinator trend data it is difficult to identify target levels.

1133 Whilst the wider pollinator community can respond positively to local habitat floral
1134 management (Heard et al., 2012; Knop et al., 2006), there have been few studies of diversity
1135 and abundance responses at the farm scale or above. This is important since agricultural
1136 intensification influences biodiversity and ecosystem processes by reducing the
1137 heterogeneity, composition and configuration of habitats at difference spatial scales
1138 (Tschardt et al., 2012). This is expected to have distinct effects on different pollinator
1139 guilds and pollination services but remains largely unexplored (Carvell et al., 2011;
1140 Holzschuh et al., 2007). There is still debate as to whether currently available AES are
1141 enhancing farmland biodiversity efficiently (Kleijn et al., 2011), especially pollinators

1142 (Scheper et al., 2013). In addition, there is concern that AES may suppress yields (e.g.
1143 through land-use costs), thereby increasing pressure on non-cropped habitats, with
1144 unintended negative environmental consequences. There is thus a clear need to understand
1145 and predict how wild pollinator community structure and functional attributes are altered by
1146 AES delivery and landscape configuration, especially if a balance must be struck between
1147 reducing the effects of multiple pressures and the wider need for more sustainable agriculture.
1148 The pollinator communities associated with different habitat types, landscapes and regions
1149 are being documented in various research projects and monitoring programmes (e.g. EU
1150 project QuESSA, UK project Agriland) are now under development (Baldock et al., 2015;
1151 Carvell et al., 2014). The relative importance of different specific resources or their spatial
1152 layout to pollinator communities could be estimated using existing ecological knowledge of
1153 many pollinators. These approaches could be combined in either highly parameterised
1154 process-based models or more generic conceptual models to explore to what extent different
1155 actions deployed in different scenarios at landscape scales meet biodiversity protection goals
1156 or alter the delivery of more sustainable agriculture (Kremen et al., 2007; Lonsdorf et al.,
1157 2009; see: www.naturalcapitalproject.org/InVEST.html).

1158

1159 **4.7 Insect pollinators in urban areas**

1160

1161 Pollinators in urban areas are relatively understudied in comparison to those in rural
1162 habitats, although interest in how urban areas might sustain pollinator populations and
1163 provide pollinator services has increased in recent years (Baldock et al., 2015; Bates et al.,
1164 2011; Bonaszak & Zmihorski, 2012; Geslin et al., 2013; Hernandez et al., 2009; Ireneusz-
1165 Hennig & Ghazoul, 2012; Sirohi et al., 2015). Pollinator services provided by insects in urban
1166 areas include pollination for urban food production (i.e. subsistence growing of vegetables

1167 and fruits) and pollination of native plant species that grow in urban areas, as well as
1168 providing aesthetic value to people desiring interaction with nature on their doorstep. Urban
1169 habitats can harbour important insect pollinator diversity, with the pollinator fauna of our
1170 towns and cities being at least as diverse as those of the surrounding countryside (Baldock et
1171 al., 2015; Sirohi et al., 2015). Considering that urbanised land cover in many countries has
1172 greatly expanded in the past few decades (for example, across UK (National Ecosystem
1173 Assessment 2011) and Europe (Gerard et al., 2010)), and that globally the proportion of
1174 people living in urban areas now exceeds 50% (UNFPA 2007), it is perhaps surprising that
1175 we know relatively little about the impact of towns and cities on insect pollinator populations,
1176 communities and networks (Geslin et al., 2013). Moreover, how should we manage urban
1177 areas most effectively to benefit pollinators? Given the recent policy interest in pollinator
1178 conservation (see section 3), practitioners require information and advice on how to manage
1179 public and private urban land to benefit pollinators. This is challenging, because perhaps
1180 more than any other habitat type, urban areas represent a matrix of different land use types
1181 (Wray and Elle, 2015), from car parks to allotments. Larger areas with more pollinator
1182 friendly habitat, such as nature reserves, parks, allotments and gardens, are connected by
1183 strips of potentially pollinator friendly habitat including roadside verges, hedgerows, river
1184 banks and canal towpaths, railway cuttings and embankments (Figure 11). Understanding
1185 how pollinator populations develop within such a ‘grainy’ landscape requires understanding
1186 of how habitat diversity, patch size and connectivity influences their meta-population
1187 dynamics (Cane et al., 2006; Perovic et al., 2015).

1188

1189 We currently have a limited understanding of whether urban areas act as sources or
1190 sinks for pollinator populations, either in terms of population numbers or genetic diversity.
1191 We know that urban habitats can support high abundance and species richness of some

1192 pollinator taxa (bees) but not others (hoverflies) relative to farmland and nature reserves
1193 (Baldock et al., 2015), yet we know very little of the connectivity of pollinator populations
1194 across habitat types. Quantifying the scale and extent of pollinator movement into and out of
1195 cities is crucial to understanding the role of urban pollinator populations in regional
1196 ecosystem service provision. This issue is also linked to the roles of landscape features that
1197 allow population dispersal – particularly linear features such as hedgerows, roadside verges
1198 and waterways. It is important to know how habitat connectivity is influenced by pollinator
1199 traits, such as levels of generality in plant associations, population structure (e.g. social vs
1200 solitary bees) and observed relationships between body size and dispersal ability (Greenleaf
1201 et al., 2007; Hagen et al., 2012). Questions associated with population structure can be
1202 addressed using genetic analyses across urban-rural gradients, and across contrasting urban
1203 habitat types, replicated in a range of towns and cities of different sizes. In addition, more
1204 mark-release-recapture studies are required for direct estimation of dispersal and foraging
1205 distances over single generation, ecological timescales (Chapman et al., 2003).

1206

1207 Little is known about the role of urban pollinators in mediating gene flow between
1208 plant populations within and outside cities (Roberts et al., 2007). This will require a greater
1209 understanding of the intricacies of pollen transport, for example by linking pollinator
1210 dispersal with estimates of gene flow (Figure 11). On a landscape scale, rare long-range
1211 pollen dispersal events may be more important for maintaining the genetic diversity of
1212 populations than frequent but local pollen dispersal (Wilcock and Neiland, 2002). We also
1213 need to understand the efficacy and impact of pollen flow between the many cultivars of
1214 native or naturalised plant species grown in urban environments and native provenances in
1215 surrounding non-urban habitats. Such gene flow has the potential to maintain genetic
1216 diversity in native populations, or could harm them through outbreeding depression or

1217 genetic assimilation (Wilcock and Neiland, 2002).

1218



1219
1220

1221 **Figure 11** Representation of an urban landscape showing a dynamic environment **a)** Can
1222 insect pollinators be permanent residents in such habitats? A matrix of different land-use
1223 types that may be deemed as pollinator friendly (e.g. allotments, gardens, or hedgerows), or
1224 unfriendly (e.g. high density industrial, commercial or housing) areas. **b)** Are features such as
1225 verges of roads and waterways (and railways) important in allowing pollinator movement to
1226 connect patches of suitable resources within cities, as well as providing corridors for
1227 pollinator and plant gene flow? **c)** Species richness in some urban areas can be comparable
1228 with some rural areas (Baldock et al., 2015), and so we need to establish if urban areas act as
1229 pollinator sinks, drawing in pollinators from surrounding rural areas, or as sources,
1230 replenishing perturbed pollinator communities in rural areas. Either way, migration between
1231 urban and rural areas may be important in mediating plant gene flow between visited flowers.
1232 **d)** Bringing aesthetic value to urban areas by: observing pollinators on your doorstep, and
1233 benefitting from their pollination service such as increasing the health of visually beautiful
1234 flowering plants in parks, allotments and gardens. **e)** Increasing interest in urban food
1235 production (such as garden beans pictured), both domestically and commercially, requires a

1236 healthy pollinator community in urban areas in a similar way to agriculture practices in rural
1237 areas.
1238 The growing of food in cities has increased dramatically in recent times; worldwide, urban
1239 agriculture involves an estimated 100 million people (Eigenbrod and Gruda, 2015), covers an
1240 area of at least 68 million hectares, and accounts for around 16% of lands devoted to crops
1241 (Thebo et al., 2014). Previous studies have demonstrated that urban agriculture can support
1242 diverse communities of bees and butterflies (Matteson et al., 2008; Matteson and Langellotto,
1243 2010), but the value of the pollination services provided by pollinators in urban environments
1244 remains unquantified (Lin et al., 2015). To understand the economic value of pollinators in
1245 urban areas we need to identify the range and abundance of crops grown in urban areas, and
1246 the role of insect-mediated pollen flow for these crops (Van Rossum, 2009, 2010).
1247 Assessment of the ecosystem service value of urban pollinators requires estimation of the
1248 economic value of their contributions both to urban agriculture, and crop production in
1249 surrounding non-urban areas.

1250

1251 Management of urban habitats for pollinators requires understanding of how to
1252 improve habitat quality, and of how to link good quality habitat patches in ways that best
1253 support pollinator populations. Habitat connectivity is crucial to the maintenance of viable
1254 populations and pollinator species richness; urban areas are thought to restrict gene flow in
1255 some bumblebees (Jha, 2015; Jha and Kremen, 2013b) and some solitary bees (Davis et al.,
1256 2010). However, it is very likely that different urban habitat types vary markedly in their
1257 impacts on dispersal and gene flow. We would expect pollinators to move more easily along
1258 flowering linear features than across large areas of flower-free man-made surfaces (though
1259 bees and butterflies will follow non-rewarding linear routes between flower patches; Cranmer
1260 et al., 2012). Flower-rich road verges, for example, not only support pollinators, but also
1261 provide potential dispersal routes, highlighting the need to develop management best

1262 practices for these and other linear features (Dicks et al., 2013; Hanley and Wilkins, 2015;
1263 Henriksen and Langer, 2013; Hopwood, 2008; Noordijk et al., 2009; Wojcik and Buchmann,
1264 2012). The growing popularity of planted wildflower meadows in cities will increase food
1265 resources for pollinators, and perhaps more than any other single effect, individual human
1266 behaviour (in terms of the way we plant and manage gardens and other green spaces, such as
1267 parks and allotments) can exert significant impact on habitat quality for plants and
1268 pollinators.

1269

1270 To date, few studies have examined the effect of management approaches on urban
1271 pollinators, and those that have been carried out focus primarily on single sites (Blackmore
1272 and Goulson, 2014; Garbuzov et al., 2015). To guide management practice, future work
1273 needs to consider societal impacts as well as effects on biodiversity. We need to understand
1274 and quantify the impact on public health and well-being from enhancing public green-space
1275 for pollinators, and need to better understand the relationships between these benefits and the
1276 cost and frequency of management interventions in urban areas (such as mowing or pesticide
1277 application). Identification of general patterns, whilst appropriately accounting for site and/or
1278 year characteristics, requires longitudinal studies monitoring insect pollinators to elucidate
1279 the impacts of alternative management regimes and interventions over multiple sites and
1280 years.

1281

1282

1283 **5. Considerations when developing future research and mitigation** 1284 **strategies**

1285

1286 A major goal in supporting insect pollinators is to get the right balance between land-

1287 use for anthropogenic activities, such as food production or urbanisation, versus the
1288 responsibility we have in conserving healthy and diverse insect pollinator populations and
1289 communities. To ensure we achieve this goal, a key approach to land management strategies
1290 is to enact evidence-based decision making rather than adopting uniform responses to
1291 political agendas. Policy initiatives designed to support excellent research to understand
1292 pollinator sustainability, with the intention of using research outputs to better inform decision
1293 making, should therefore be both applauded and supported. However, the challenges put
1294 forward to the research community often comprise very broad questions and goals that may
1295 not be straightforward to tackle or simple to implement. It is important that researchers
1296 communicate and feedback with both clarity and transparency about their intended and
1297 ongoing research, so that objectives can be developed in ways that are useful for researchers,
1298 policy and practice, as by doing so any mitigation actions requiring development can be
1299 achieved more rapidly and effectively.

1300

1301 It is also important that we align research developments by considering the concerns
1302 raised by stakeholders and identify areas which are suffering (or are at future risk) from
1303 pollination deficits. Consequently, researchers should look to co-develop proposals with the
1304 right balance of all stakeholders, from academic, conservation, industry, policy and
1305 practitioner backgrounds, with their needs considered in order to apply research to the
1306 practical world (Dicks et al., 2013; Turnhout et al., 2012). For instance, a common criticism
1307 is the ambiguity of the term, ‘ecosystem service’, when describing the process of pollination.
1308 When researchers use this term, it is important they consider the distinct difference between
1309 the study of ecosystems functions and ecosystem services, by considering how their research
1310 (for example on insect pollinators) also provides a direct or indirect socio-economic benefit,
1311 and how actions might then be implemented by land managers to deliver pollination services,

1312 especially in the specific case of improving flowering crop production or the health of wild
1313 flower populations. Furthermore, although understanding the specific risks posed to insect
1314 pollinators is no doubt crucial, it is only one piece of the whole puzzle. The remaining pieces
1315 required are to understand the actions required to mitigate such threats. To achieve this we
1316 require knowledge about the likely uptake of certain schemes and the behaviours of end-users
1317 that hopefully results in them taking action. Research needs to thus be tailored towards
1318 informing effective management to better inform viable management of the land to support
1319 insect pollinators. This latter point is critical because we need to consider wider values, such
1320 as how we might best meet the needs of farmers / land owners in rural areas, or perhaps
1321 planners, green space designers and residents in urban areas and consider providing the
1322 aesthetic and cultural values to the public.

1323

1324 Kennedy et al (2013), suggested that for each 10% increase in area of high-quality
1325 habitat (for bees), for instance a conversion of intensively farmed agricultural land to a more
1326 natural habitat land-use, could return around a 37% increase in species abundance and
1327 richness (for wild bees). Achieving this goal, however, may prove difficult because economic
1328 interests may conflict with conserving suitable habitats which is an issue of high relevance to
1329 large scale agriculture that relies on efficient land-use. Land managers appear to be relatively
1330 open to discussing the potential of managing land for pollinators, but they are looking for
1331 clear, concise and balanced advice about how to most effectively do this with as little cost
1332 and time incurred as possible. Thus, to most effectively deliver advice we must be
1333 considerate of the pressures placed on them. Farmers can be classed as custodians of the
1334 countryside, yet they are also businessmen and are under pressure to make a profit whilst
1335 meeting consumer demands for the countryside. For farmers to heed to specific advice,
1336 therefore, we should be attempting to provide a demonstrable, specific action, applicable to

1337 their businesses (a ‘true’ not hypothetical ecosystem service). If we perceive an action as
1338 being an ecological enhancement of the land (in this case increasing stable pollinator
1339 communities and / or pollinator diversity) we need to make this relatively cost effective and
1340 make the link clear to how such an ecosystem service can provide potentially economic and /
1341 or even aesthetic enhancement. Similarly, the same principle approach should be taken when
1342 encouraging the general public to support insect pollinators, for instance, in designing garden
1343 habitats to support a diverse set of insect pollinators.

1344

1345 It is of further importance that the implementation of specific actions considers what
1346 the short versus long-term effects will be (Blaauw and Isaacs, 2014a). For example, when
1347 engineering habitats to help conserve insect pollinators, how long might it take for an
1348 agricultural field margin or urban flower bed sown with a ‘pollinator friendly’ plant seed mix
1349 to establish? How long does this plant community remain in flower and does the composition
1350 of flower species change over time? How long does the floral community remain attractive to
1351 insect pollinators, and how regularly does it need to be managed to keep it free of invading
1352 non-flowering plants? As we have discussed, concerted research efforts and knowledge
1353 exchange to determine and recommend the best implementation methods are thus required
1354 (Arlettaz et al., 2010), but this must be followed up with close monitoring to determine
1355 whether such recommendations actually benefit target insect pollinators (Stem et al., 2005).

1356

1357

1358 **Contribution**

1359

1360 The order of author appearance broadly depicts the level of contribution to the paper:
1361 RJG administered the development, organised and put together the paper; RJG and SGP

1362 initiated the collaboration; RJG, KCRB through to GW, and SGP contributed to conceiving
1363 ideas, acted as champions of identified evidence gaps and were involved in writing the paper;
1364 ANA provided significant input into putting the paper together; ANA through to PBS
1365 contributed to conceiving ideas and comments on the paper. Figures were conceived and
1366 produced by RJG, LAG, JMH, CQT and ANA.

1367

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Appendix

Table 2: Selected global, regional and national initiatives relevant to pollinators and pollination service.

	Initiative name	Scale	Description
1	Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES)	Global	Independent intergovernmental body established to strengthen the science-policy interface for biodiversity and ecosystem services conservation and sustainable use of biodiversity, long-term human well-being and sustainable development. The “Pollinators, pollination and food production” assessment aims at enhancing policy responses to pollinator declines, and deficits in pollination.
2	Convention on Biological Diversity (CBD)	Global	CBD programme on agricultural biodiversity has an International Initiative for the Conservation and Sustainable Use of Pollinators. In addition the CBD Aichi Biodiversity Targets cover pollinators and pollination services.
3	International Pollinator Initiative (IPI)	Global	International Initiative for the Conservation and Sustainable Use of Pollinators Plan of Action consists of four basic elements: Assessment, Adaptive Management, Capacity Building, and Mainstreaming. FAO has been coordinating and facilitating the implementation of the IPI by undertaking, in collaboration with numerous partners, activities that contribute to the implementation of these four elements.
4	Sustainable Development Goals (SDGs)	Global	SDGs will replace the Millennium Development Goals with a change in paradigm with biodiversity now having its own goal. Goal 15: “Protect, restore and promote sustainable use of terrestrial ecosystems...and halt all biodiversity loss” which will help underpin Goal 2: “End hunger, achieve food security and adequate nutrition and promote sustainable agriculture”.
5	Global Pollination Project	Global	Global Environment Facility (GEF) funded project to consolidate the knowledge base to inform good agricultural practices to improve food security, nutrition and livelihoods through enhanced conservation and sustainable use of pollinators (Brazil, Ghana, India, Kenya, Pakistan, Nepal and South Africa). The project is coordinated by the Food and Agriculture Organization of the United Nations, with implementation support from the United Nations Environment Programme (UNEP).
6	Prevention of honeybee Colony Losses (COLOSS)	Global	COLOSS is an international, non-profit association headquartered in Bern, Switzerland focussed on improving the well-being of bees at a global level. COLOSS is composed of scientific professionals incl. researchers, veterinarians, agriculture extension specialists, and students from 69 countries.
7	Global Biodiversity Information Facility (GBIF)	Global	GBIF is an international open data infrastructure, funded by governments that allows anyone, anywhere to access data about all types of life on Earth, incl. many pollinator species, shared across national boundaries via the web.
8	International Commission for Pollinator Plant Relationships (ICCP)	Global	ICCP has the following objectives to: (i) Promote and coordinate research on relationships between plants and bees; (ii) Organise meetings, colloquia or symposia related to the above topics, and to publish and distribute the proceedings; (iii) Collaborate closely with national and international institutions interested in the relationships between plants and bees, particularly those whose objectives are to expand scientific knowledge of animal and plant ecology, and fauna protection.
9	Status and Trends of European Pollinators (STEP)	Regional (Europe)	Large scale research project funded by the EC which is assessing the status and trends of pollinators in Europe, quantifying the relative importance of various drivers and impacts of change, identifying relevant mitigation strategies and policy instruments, and disseminate this to a wide range of stakeholders.
10	Sustainable pollination in Europe - joint research on bees & other pollinators (SUPER-B)	Regional (Europe)	SUPER-B is a knowledge exchange network bringing together scientific and societal communities involved in the conservation and sustainable management of ecosystem services mediated by pollinators

11	North American Pollinator Protection Campaign (NAPPC)	Regional (N. America)	NAPPC's mission is to encourage the health of resident and migratory pollinating animals in N. America to: (i) Raise public awareness and education and promote constructive dialogue about pollinators' important to agriculture, ecosystem health, and food supplies; (ii) Encourage collaborative, working partnerships among participants and federal, state and local government entities. This will strengthen the network of associated organizations working on behalf of pollinators; (iii) Promote conservation, protection and restoration of pollinator habitat; (iv) Document and support scientific, economic and policy research - creating the first-ever, international data bank (library) of pollinator information
12	African Pollinator Initiative (API)	Regional (Africa)	API has two main purposes to: (i) facilitate African country participation in the International Pollinator Initiative's (IPI) global pollinator project; and (ii) improve pollinator biodiversity conservation, and the pollination of crops and wild plants through networking.
13	Oceania Pollinator Initiative (OPI)	Regional (Oceania)	Key functions of the OPI are to: (i) 1. Monitor pollinator decline, its causes and its impact on pollination services; (ii) Address the lack of taxonomic information on pollinators; (iii) Assess the economic value of pollination and the economic impact of any decline; (iv) Promote conservation, restoration, and sustainable use of pollinators in agriculture and ecosystems.
14	England's National Pollinator Strategy (NPS)	National (England, UK)	NPS vision is to see pollinators thrive, so they can carry out their essential service to people of pollinating flowers and crops, while providing other benefits for our native plants, the wider environment, food production and overall human welfare.
15	Brazilian Pollinator Initiative (BPI)	National (Brazil)	BPI focuses on national policies, research, scientific meetings and learning activities. A national evaluation on pollinator and pollination status in Brazil is under construction, and strategies and actions are underway to consolidate the Brazilian Pollinators Network.
16	White House – Pollinator Research Action Plan	National (USA)	As part of a larger Federal strategy to ensure pollinator health, President Obama issued the Presidential Memorandum “Creating a Federal Strategy to Promote the Health of Honey Bees and Other Pollinators.” The memorandum called on the Federal government to draft a pollinator research action plan
17	Wales Action Plan for Pollinators	National (Wales, UK)	Welsh Government has worked with industry and stakeholders to look in more detail at the evidence and issues around pollinators and their conservation in Wales. Following consultation the Action Plan for Pollinators sets the strategic vision, outcomes and areas for action to improve conditions for pollinators and work to halt and reverse their decline in Wales.
18	Canadian Pollination Initiative (CANPOLIN)	National (Canada)	CANPOLIN was a large scale research project addressing the growing problem of pollinator decline in agricultural and natural ecosystems in Canada.
19	French National Action Plan	National (France)	A ten point action plan for pollinators
20	Insect Pollinators Initiative (IPI)	National (UK)	UK research funders invested £10m in an Insect Pollinator Initiative which supported multiple research projects aiming to understand the causes and consequences of threats to insect pollinators.